Greenhouse Gas Balance for Composting Operations

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The greenhouse gas (GHG) impact of composting a range of potential feedstocks was evaluated through a review of the existing literature with a focus on methane (CH_4) avoidance by composting and GHG emissions during composting. The primary carbon credits associated with composting are through CH_4 avoidance when feedstocks are composted instead of landfilled (municipal solid waste and biosolids) or lagooned (animal manures). Methane generation potential is given based on total volatile solids, expected volatile solids destruction, and CH_4 generation from lab and field incubations. For example, a facility that composts an equal mixture of manure, newsprint, and food waste could conserve the equivalent of 3.1 Mg CO_2 per 1 dry Mg of feedstocks composted if feedstocks were diverted from anaerobic storage lagoons and landfills with no gas collection mechanisms. The composting process is a source of GHG emissions from the use of electricity and fossil fuels and through GHG emissions during composting. Greenhouse gas emissions during composting are highest for high-nitrogen materials with high moisture contents. These debits are minimal in comparison to avoidance credits and can be further minimized through the use of higher carbon:nitrogen feedstock mixtures and lower-moisture-content mixtures. Compost end use has the potential to generate carbon credits through avoidance and sequestration of carbon; however, these are highly project specific and need to be quantified on an individual project basis.

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Concerns over climate change as a consequence of the release of greenhouse gases (GHGs) has resulted in a range of efforts to regulate and reduce their emissions and to replenish the stores of fixed carbon (C) on earth. As a result of these efforts, there is now a financial value and international markets for stored C or C/GHGs that are not released into the atmosphere. Expected increases in the stringency of regulatory frameworks will likely lead to increased values for C, making C-offset projects more financially appealing. As with other commodities, accounting systems have been developed in an attempt to quantify changes in C emissions associated with different practices. Greenhouse gas accounting is done by evaluating the debits and credits associated with a particular practice. Debits are emissions of GHGs into the atmosphere, and credits are essentially deposits of C into a fixed, stable form. There may also be credits associated with emissions avoidance of GHGs as a result of a change in standard practice.

Considering the GHG impact of a practice that is generally considered environmentally beneficial has the potential to increase or decrease the benefits associated with the practice. Composting is one such practice. Composting is an aerobic process that transforms a range of organic substrates into a stable, humus-like material through microbial decomposition. Composting can be considered to be a C-based system, categorically similar to reforestation, agricultural management practices, or other waste management industries. Unlike "smoke-stack" industries, which are relatively simple to document, biologically based C credits are inherently more variable. Because of this, accurate GHG accounting and best management practice standards will likely become an important consideration in terms of the "quality" of C credits and market risks and values associated with C trading for these credits. Additionally, incorporation of the best available information for determining potential credits associated with different biologically based C systems will lend credibility to C credits or debits associated with these systems.

By most indices, including the USEPA's evaluation of waste, composting is considered an environmentally friendly practice (USEPA, 2002). The feedstocks used for composing are often residuals or wastes from a range of industries that are often diverted from the solid waste

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Abbreviations: COD, chemical oxygen demand; GHG, greenhouse gas; HRT, hydraulic retention time; MSW, municipal solid waste; TVS, total volatile solids.

stream. Because these feedstocks are part of the short-term C cycle, the CO_2 emissions from decomposing organic matter in compost piles are not considered as additional GHG emissions. In addition, by composting these materials, a stable, reduced pathogen soil conditioner that may have nutrient value is produced from materials that have potentially been diverted from storage lagoons and landfills. Most composting operations are likely to function as a source of GHGs and as a means to avoid GHG release at different stages of their operations. The stages of a composting operation that have the potential to affect GHG emissions include selection of feedstocks, transport to and from the compost site, energy use during composting, gas emissions during composting, and end uses of compost products. The debited portions of the composting process include transport, energy use during composting, and fugitive emission of GHGs other than CO_2 from composting operations. The credited portions of the composting process include diversion of feedstocks from storage or disposal where they would generate CH_4 as well as end use of compost products.

The greenhouse gases that have the potential to be emitted by compost operations include CH_4 and N_2O. Methane is formed as a by-product of microbial respiration in severely anaerobic environments when C is the only electron acceptor available. Carbon is used as an electron acceptor when other, more energetically favorable electron acceptors, including oxygen, nitrogen, iron, manganese, and sulfur, have been exhausted. Because the environments in a waste storage lagoon, landfill, or compost pile are not uniform, it is also possible that different electron acceptors can be used simultaneously. For example, when sulfur is used as an electron acceptor, highly odorous compounds, including dimethyl disulfide and methyl mercaptan, are formed. The presence of these compounds can be indicative of the presence of CH_4. A compost or waste pile that exhibits minimal odors is more likely to have aerobic conditions throughout than a malodorous pile.

The conditions favorable to the formation of N_2O are not as well understood (Béline et al., 1999). Nitrous oxides can be formed during nitrification (conversion of NH_3 to NO_3^-) and denitrification (conversion of NO_3^- to N_2) reactions, although they are much more commonly associated with denitrification (Brady and Weil, 2001). Denitrification involves the following transformations:

$$2\ NO_3^- \rightarrow 2NO_2^- \rightarrow 2\ NO \rightarrow N_2O \rightarrow N_2 \qquad [1]$$

In soil systems, N_2O is much more likely to form and volatilize when the carbon to nitrogen (C/N) ratio of soil organic matter is low (<20:1), and denitrification is much more likely to occur under anaerobic conditions (Brady and Weil, 2001; Huang et al., 2004; Klemedtsson et al., 2005). For some waste storage areas or composting operations, the C/N ratio of the material is high enough and/or the mixture is dry enough to limit denitrification reactions.

A number of recent studies have evaluated the GHG credits and debits of composting (Clean Development Mechanism, 2006; Recycled Organics Unit, 2006; Smith et al., 2001; USEPA, 2002; Zeman et al., 2002). For emissions associated with composting, each of these considered energy requirements for transport of feedstocks to the compost facility and finished product to the end user as debits associated with the process. Energy use during composting was also included. Two considered a turned windrow composting system for their model, whereas Smith et al. (2001) also considered closed systems with energy requirements for odor control (Recycled Organics Unit, 2006; USEPA, 2002). Only Zeman et al. (2002) and Clean Development Mechanism (CDM) (2006) considered fugitive emissions of GHG, such as CH_4 or N_2O, as debits. Two (Recycled Organics Unit, 2006; USEPA, 2002) used yard waste as the sole feedstock, whereas Smith et al. (2001) considered organics diverted from municipal solid waste (MSW), which includes food and yard waste. For GHG credits associated with composting, each of the reviews used different scenarios. The Recycled Organics Unit (2006) and the USEPA (2002) considered different agricultural end uses, whereas Smith et al. (2001) considered the potential for compost to replace peat for a range of applications. The United Nations Framework Convention on Climate Change, Clean Development Mechanism (CDM, 2006) was the only study to include avoidance of methane (CH_4) generation by the aerobic decomposition of feedstocks in a compost pile in comparison to the anaerobic breakdown of these materials in other disposal sites in the credits associated with composting.

This review was conducted to quantify potential GHG debits and credits associated for a wide range of feedstocks and composting systems. We break the composting process into three distinct phases: (i) the CH_4 generation potential of different compost feedstocks; (ii) the composting process, including energy use and fugitive emissions during composting; and (iii) the end use of compost. The first phase describes the GHG generation potential of different feedstocks. This is pertinent for composting operations that divert feedstocks from disposal sites or storage facilities where they would be expected to generate CH_4. Because anaerobic digestion is a common practice for municipal biosolids and is coming to be recognized as a viable stabilization technology for other types of wastes, efficiencies of digestion and remaining CH_4 generation potential, post digestion, are also examined. Anaerobic digestion with CH_4 capture for energy use or flaring are important techniques for a source of energy and for avoidance of fugitive GHG release (Matteson and Jenkins, 2007). Second, we detail the potential for gas generation for during the composting process. The literature on gas emission potential for different feedstocks is summarized for storage before composting and during the composting process. Different process variables and their effect on gas emissions are also discussed. Energy requirements for different compost systems are summarized. Finally, potential end uses of compost and their impact on GHGs are briefly discussed. Complications associated with credits and debits for this stage of the process are described. The implications of compost end use on GHGs are qualitatively assessed. This review is intended to be sufficiently detailed so that credits and debits associated with specific composting operations can be determined.

Compost Feedstocks

Materials that are commonly used as feedstocks for composting operations include food, paper, and yard trimmings that are separated from the municipal solid waste stream, municipal biosolids, and animal manures. For each of these materials, conventional disposal practices are associated with the release of CH_4 into the atmosphere. Food, paper, and yard trimmings are generally landfilled. Biosolids can also be landfilled (often after anaerobic digestion). Food is landfilled and is also a component of biosolids through the use of under-sink grinders. Emissions of CH_4 from landfills are expected to increase as the quantity of waste generated increases with increasing prosperity. One estimate shows emissions from MSW increasing from 340 Mt CO_2eq in 1990 to 2900 Mt CO_2eq in 2050 (Bogner et al., 2007). In the USA, animal manures are commonly stored in uncovered lagoons. According to a recent United Nations publication, storage of animal manures in lagoons or liquid holding tanks generates up to 18 million Mg of CH_4 per year (Steinfeld et al., 2006). The CH_4 generation potential of each of the compost feedstocks is detailed in the next section. This potential is intended to reflect the maximum quantity of CH_4 that can be released through anaerobic decomposition. Other site-specific factors may influence the amount of CH_4 that is released into the atmosphere as a result of decomposition. Some examples of these factors include landfill cover soil oxidation of CH_4 and reduced decomposition due to cold temperatures or lack of moisture (Brown and Leonard, 2004ab). Some landfills have extensive landfill gas collection systems, although there is a great deal of variability in reported collection efficiencies (Themelis and Ulloa, 2007). The values presented in this review can be modified to reflect site-specific conditions.

Manures and Biosolids

Methane Generation Potential

For biosolids and manures, CH_4 generation potential is generally expressed in relation to the total volatile solids (TVS) content of the feedstock (Metcalf and Eddy, 2003; IPCC, 2006). The TVS is defined as the portion of the carbonaceous material that volatilize on ignition at 500°C. A certain portion of the TVS readily decomposes and forms biogas under anaerobic conditions. The remainder is generally considered refractory and may decompose under aerobic conditions over time. The portion of the TVS that readily decomposes can be directly related to the CH_4 generation potential of the feedstock. An equation was developed to predict the gases that will evolve from anaerobic decomposition of different feedstocks based on the chemical composition of the feedstock (Metcalf and Eddy, 2003):

$$\begin{aligned} C_vH_wO_xN_yS_z &+ (V - w/4 + x/2 + 3y/4 + z/2) \times H_2O \\ &\rightarrow (v/2 + w/8 + x/4 + 3y/8 + z/4)CH_4 \\ &+ (v/2 + w/8 + x/4 + 3y/8 + z/4)CO_2 \\ &+ yNH_3 + zH_2S \end{aligned} \qquad [2]$$

The Intergovernmental Panel on Climate Change (IPCC) has identified values for CH_4 generation based on TVS and the maximum CH_4 production capacity for different types of livestock (IPCC, 2006). Here, the importance of diet in determining the TVS content and the CH_4 generation potential of the manure is emphasized. Different values are provided for different continents, and collection of site-specific information is recommended. Table 1 shows volatile solids produced per animal per day, the portion of the volatile solids (VS) that are easily decomposed and the resulting energy value or CH_4 generation potential (Table 1). The table has been expanded to include the CO_2 equivalence of the CH_4 per day and per year and comparative values from the IPCC for animals raised in North America. The values from this publication are similar to those used for swine, feedlot, and dairy cattle in another study (Garrison and Richard, 2005). The IPCC guidelines specifically recommend the use of locally generated data because feed type, specific breed, and local climate factors affect CH_4 generation potential of the liquid wastes. In this case, the values for dairy cows in Hansen (2004) are higher than the IPCC values, whereas the values for poultry are lower than the IPCC values. Factors that influence values on a regional level can be animal and climate specific. Animal breed and diet effect waste properties. Cooler ambient temperatures can result in up to a 50% decrease in CH_4 production in cooler regions (average annual temperature ≤10–14°C). This decrease is meant for outdoor lagoons where temperature within the lagoon is not controlled. The IPCC guidelines discount the potential for N_2O evolution from uncovered liquid or slurry storage or liquid waste lagoons. The information in Table 1 can be used to estimate the maximum CH_4 value (or its CO_2 equivalent) for different types of animal manure.

Manures are often stored or lagooned before composting. Decomposition occurs during storage. The IPCC has a protocol for CH_4 avoidance credits for manure lagoons that are covered to prevent gas release. The credits are based on the CH_4 generation potential of the manures, specific climatic factors, and the residence time for materials in the lagoons (Dong et al., 2006). For differences between fresh and stored manure, the IPCC guidelines specify appropriate reduction for factors including type and length of storage (Dong et al., 2006). Some of these values are based on published literature, whereas others are based on the judgment of the authors. For example, potential CH_4 conversion factors for uncovered lagoons range from 66% to 80% depending on temperature. Actual CH_4 evolution from lagoons varies by season, location, and lagoon design (DeSutter and Ham, 2005; Dong et al., 2006; Shores et al., 2005). In lieu of actual data, it is also possible to use the default values provided in the IPCC doc-

Table 1. Volatile solids (VS) produced per 455 kg animal unit per day, likely VS destruction, total gas, CH_4 produced (using 22.4 L per mole of CH_4), and CO_2 equivalent for a range of animal wastes (Hansen, 2004).

	VS per animal per day	Likely VS destruction	Total gas	CH_4 d^{-1}		CO_2 equivalent		IPCC Value
	kg	%	m^3 d^{-1}	m^3 of CH_4	Moles CH_4	kg d^{-1}	——Mg yr^{-1}——	
Beef	2.68	45	0.84	0.50	22.5	8.28	3.02	
Dairy	3.91	48	1.23	0.74	33	12.14	4.43	1.1–2.6
Swine	2.18	50	0.81	0.49	21.75	8	2.92	1.9–4.3
Poultry layers	4.27	60	2.02	1.21	54	19.9	7.25	12.6–14.7
Poultry broilers	5.45	60	2.58	1.55	69	25.4	9.27	

ument for appropriate reduction factors. In addition, a measure of the VS of the lagooned or stored manure may be an indication of the amount of stabilization that has taken place and the subsequent loss of CH_4 generation potential.

Anaerobic Digestion

Municipal biosolids and animal manures may have undergone anaerobic digestion for stabilization and pathogen reduction (biosolids) or for energy recovery (manures) before composting. For biosolids, retention time in digesters is typically determined by volatile solids and pathogen reduction. Here, retention time in the digester may be less than required for complete CH_4 recovery. In the case of animal manures, retention time in digesters is calculated to maximize CH_4 generation and minimize storage time, a decision based on economic optimization. This suggests that if the manures are returned to piles or lagoons after digestion, additional CH_4 or N_2O can evolve from the storage areas. The difference between maximum and actual CH_4 generation in anaerobic digesters can be viewed as the remaining CH_4–generating potential of the substrate. Data from different studies support this concept. For example, Clemens et al. (2006) looked at CH_4 generation for raw and anaerobically digested cattle slurry stored in open and closed containers. Although CH_4 generation was highest for the raw slurry (36 kg CO_2 eq m^{-3}), digested slurry (29 d hydraulic retention time [HRT] in the digester) also emitted significant amounts of CH_4 (15 kg CO_2 eq m^{-3}).

There are several types of anaerobic digesters that are used in farming operations. These include covered lagoons, complete mix digesters, plug-flow digesters, and fixed film digesters. Each of these systems has a different optimal performance potential. Covered lagoons have an average residence time of 40 d and require the least management. Complete mix digesters treat manure with total solids content of 3 to 10% and a residence time of 15 d. Plug flow digesters can handle materials with higher solids content (11–13%) with a moderate retention time, usually between 20 and 25 d. Fixed-film digesters are appropriate for material with very low solids content and for short retention times (Kruger, 2005). Reported residence times in lagoons and complete mix digesters are averages, and the actual residence time of any given substrate sample is highly variable. In addition, actual CH_4 generation within each type of system is often significantly different from predicted generation.

Factors Influencing Methane Generation

Differences between predicted and actual performance can be significant. This is illustrated with examples from studies conducted to optimize digester performance. Karim et al. (2005) looked at different types of mixing in lab-scale digesters combined with different solids content of the feedstock. Cow manure slurry at 5% solids content released up to 30% more biogas with mixing. Mixing can facilitate the distribution of methanogenic bacteria in the digester and may improve the recovery of CH_4 gas bubbles suspended in the substrate. Ben-Hasson et al. (1985) used manure with a higher solids content and found that mixing decreased CH_4 production by up to 75%. Temperature also affects the rate of CH_4 production. Nielsen et al. (2004) compared two-stage anaerobic digestion with thermophillic temperature regimes. They saw up to 8% higher CH_4 yield and 9% greater TVS reduction with the two-stage process compared with digestion at a single temperature. Bonmati et al. (2001) observed a 24 to 56% improvement in CH_4 yield with a two-stage temperature reactor using pig slurry as a feedstock.

In a report prepared for the AgSTAR program in USEPA, the performance of a mesophilic plug flow digester at a dairy farm was monitored for 1 yr (USEPA, 2005). During the course of the year, the dairy went from 750 head to 860 head. Per capita CH_4 production decreased with the increase in numbers, going from 2.08 m^3 CH_4 to 1.71 m^3 CH_4 per animal per day. This decrease shows that the digester efficiency decreased with increasing number of animals due to the reduction in actual residency time caused by the increase in loading rate. It also suggests that the CH_4 generation potential for material coming out of the digester increased. Analysis of the manure from the farm found that 47% of the TVS were degradable. Total volatile solids destruction in the digester averaged 39.6%, suggesting that the digester operated at 84% efficiency. The authors note that the digester was designed for a 22-d retention time. Actual retention time during the course of the year averaged 29 d. This extended time in the digester was partially responsible for the observed efficiency. If retention time had been limited to 22 d as specified (which would happen if the dairy further increased its herd size), efficiency may have decreased to only 33.7% destruction of TVS. The authors report that a similar study done at another dairy showed only 30% destruction of TVS. Another study reported CH_4 generation per cow per day in a dairy farm in Minnesota of 1.46 m^3 CH_4 (Goodrich et al., 2005). This was generated in a plug-flow digester, which serviced 800 head. The digester for this farm was producing 30% greater CH_4 than had been predicted. Total volatile solids and retention time in the digester were not reported.

These examples support the notion that predicted and actual performance are generally different. The difference between the estimated portion of the TVS in the raw material that is decomposable and TVS in the digested material can be used as a basis to determine the CH_4 generation potential of anaerobically digested animal manures.

Biosolids

Anaerobic digestion is used in municipal wastewater treatment for stabilization and pathogen destruction (Han et al., 1997). Sizing of digesters is based on expected solid retention time and HRT. These parameters are determined in conjunction with the design of the remainder of the treatment plant. Digesters are generally fed primary sludge from settling tanks and secondary or waste-activated sludge from biological treatment. The material from biological treatment consists of microbial biomass and can be resistant to decomposition in conventional digesters (Han et al., 1997). If a plant is operating at close to design capacity, retention time in digesters is kept at the minimum required to meet regulatory requirements. As with manure digesters, the longer the retention time, the higher the % destruction of TVS and, correspondingly, the more CH_4 that is

Table 2. Volatile solids destruction as a function of digestion time for biological waste materials undergoing anaerobic digestion at mesophillic temperatures. Data are from Metcalf and Eddy (2003).

Digestion time	Volatile solids destruction
d	%
40	69.4
30	65.5
20	60
15	56
10	50

generated during digestion. Metcalf and Eddy (2003) give approximate values for volatile solids destruction as a function of digestion time for high-rate, complete-mix mesophilic anaerobic digestion (Table 2). This paper also reports that retention times normally range from 10 to 20 d for high-rate digesters.

Volatile solids destruction is also described in Metcalf and Eddy (2003) using the following equation:

$$Vd = 13.7 \ln(\text{Solids retention time [days]}) + 18.9 \qquad [3]$$

Using this equation, a 100-d retention time would result in 82% reduction in VS. This is an appropriate value to use for maximum CH_4 generation because increased production over this time frame is minimal. Methane generation capacity of the sludge can also be estimated. If the chemical formula of the volatile solids into the digester is taken to be $C_{12}H_{22}O_{11}$, then the quantity of CH_4 will be proportional to the % reduction in VS times volume of decomposable material in the digester. The concentration of decomposable material in the liquid going into the digester depends on the type of wastewater treatment plant. Using the chemical composition of the waste given here, the mass of C converted in relation to the total VS reduction is then 42% of the total. A typical digester produces gas that is 55 to 70% CH_4, with the remainder of the C present as CO_2.

In Tacoma, Washington, sludge at the municipal wastewater treatment plant is initially treated in an aerobic thermophilic (45–65°C) digester and then transferred to an anaerobic mesophilic digester. Volatile solids at the beginning of this process are typically 80%. Anaerobic digestion eliminates 64% of the VS present at the beginning of the digestion process (Dan Thompson, Environmental Services, Wastewater, City of Tacoma, personal communication). At King County, Washington, the feed into the digester contains about 85% VS. With a retention time of 20 to 30 d, generally 65% of the incoming VS are destroyed. The incoming VS and the % destruction are considered high for what would be typical for the wastewater treatment industry (John Smyth, Wastewater Treatment Division, King County, WA, personal communication). Based on retention time in digesters and VS reduction, such as manures, the biosolids may have significant CH_4 generation potential.

As with manures, lab studies have confirmed that for biosolids, predicted CH_4 generation potential is not exhausted after standard retention times in full-scale mesophilic digesters. In a laboratory study, changing the solids retention time from 24 to 40 d in a mesophilic digester resulted in an increase in VS reduction from 32 to 47% (Han et al., 1997). In another study, sludges were taken from mesophilic digesters with retention times of 30 and 65 d (Parravicini et al., 2006). These materials were incubated in lab-scale anaerobic digesters. After 14 d, volatile suspended solids degradation efficiency was improved for both materials by 12%. Changes in digester temperatures may also increase gas generation. The most commonly used digester temperature design is single-stage mesophilic digestion. A series of studies found that an initial thermophilic stage combined with mesophilic digestion improved VS destruction with a subsequent and parallel increase in CH_4 generation (Harris and Dague, 1993; Kaiser and Dague, 1994; Han et al., 1997). Methane production per gram of VS destroyed was almost identical for a single-stage mesophilic and a combined thermophilic and mesophillic system. This confirms a relationship between VS destruction and CH_4 generation.

Food, Paper, and Yard Trimmings

Food, paper, and yard trimmings materials have traditionally been landfilled as components of MSW (U.S. Congress Office of Technology Assessment, 1989). These materials can also be used as feedstocks for compost. Within a landfill, food, paper, and yard trimmings decompose and generate CH_4. Depending on the rate of decomposition and the quantity of CH_4 associated with the decomposition, it is possible that a high percentage of the CH_4 generation potential of the substrate will be exhausted before landfill gas capture systems are put into place. Landfills constructed in the USA after 1991 with total capacities greater than 2.5 million m^3 are required to install gas collection systems to reduce CH_4 emissions (USEPA, 1999). Gas collection systems must become operative at active faces of the landfill that have been receiving waste for more than 5 yr. Closed areas must have functional collection systems beginning 2 yr after the first waste has been placed in the cell (USEPA, 1999). However, a recent study showed that CH_4 emissions from the active face of a landfill were 1.7 higher than from the sides of the fill, indicating that CH_4 generation commences before cell closure (Lohila et al., 2007). The USEPA has a model to assist solid waste professionals in calculating the total GHG emissions from decomposition of organics in landfills. The Waste Reduction Model gives single point values for gas generation potential that does not take into account the rate of decomposition of the substrate and how that might interface with the gas system requirements for larger landfills (USEPA, 1998b). In a study of the GHG potential of different end use/disposal options for MSW, Smith et al. (2001) noted that, like USEPA, the IPCC treats CH_4 emissions from landfills as though they are instantaneous. However, since that report was prepared the Clean Development Mechanism has developed a protocol for compost operations that includes a credit for CH_4 avoidance (Clean Development Mechanism, 2006). In this model, decay rates are based on a first-order decay model that accounts for different CH_4 generation potential as well as the fraction of degradable C for different waste materials. Methane generation for each material is calculated on a yearly basis until the gas generation potential for the substrate is exhausted. Wood and paper products are classified as slowly degrading, non-food organic putrescible garden and park wastes are classified as moderately degrading, and food wastes are considered rapidly degrading.

For the organic component of MSW, the CH_4 generation potential can be used as a basis for determining the CH_4 avoidance credits that would be associated with composting these materials (IPCC, 2006). Values for CH_4 generation potential of these materials in landfill environments and in anaerobic digesters has been reported along with approximate time required for decomposition (e.g., Eleazer et al., 1997). Anaerobic digestion of MSW is an established technology in Europe and is being recognized as a viable alternative to landfilling in the USA (DiStefano and Ambulkar, 2006; Zhang et al., 2007).

In laboratory scale landfills, Eleazer et al. (1997) measured CH_4 yields for grass, leaves, branches, food scraps, and an assortment of different paper grades (Table 3). The reaction vessels were maintained within the mesophilic temperature range, and water was added to assure saturation. Samples were incubated until concentrations of CH_4 were below detection limits. Observed yields were 144.4, 30.6, 62.6, and 300.7 mL CH_4 per dry g, respectively, for the grass, leaves, branches, and food scraps. Methane generation by paper products ranged from 217.3 mL CH_4 per dry g for office paper to 74.3 mL CH_4 per dry g for old newsprint. Approximate time required for decomposition, CH_4 generated per kg, and CO_2 equivalent (Mg CO_2 per Mg waste) for each waste material are shown in Table 3. These conditions were close to optimal for CH_4 generation in an unperturbed system particularly for the drier materials. In an actual landfill, moisture content of certain materials, such as food and yard trimmings, are likely to be high enough to facilitate anaerobic decomposition. Other studies have examined CH_4 yield and decomposition rate of food and yard trimmings in anaerobic digesters and septic tanks with generally higher rates of decomposition and CH_4 generation (Luostarinen and Rintala, 2007; Mähnert et al., 2005; Nguyen et al., 2007; Zhang et al., 2007). Methane generation and rate of decomposition for food scraps in anaerobic digesters are shown in Table 4.

As was the case with manures and biosolids, VS has been used to predict CH_4 generation potential for decomposition of landfilled wastes. Zhang et al. (2007) reported 81% VS destruction of food waste after 28 d in an anaerobic digester with a subsequent CH_4 yield of 435 L CH_4 kg^{-1}. In another study, higher CH_4 yields from anaerobic digestion of food waste (489 L CH_4 kg^{-1}) were attributed to a higher VS content of the feedstock (Cho and Park, 1995). Eleazer et al. (1997) reported the VS of the feedstocks but did not report %VS destroyed during the incubation. However, in another study conducted at the same laboratory using food waste, the waste (149.3 dry g) was mixed with seed material from a digester (531 dry g) (Wang et al., 1997). The seed mix had TVS of 42.2%, and the food waste had TVS of 92%. The % VS of the initial mixture was 53%. After incubation, TVS was reduced to 36.9%, which represents a 30% destruction of VS. However, CH_4 generation by the seed mix was only 5.8 mL per dry g. These results indicate close to complete destruction of the VS from the food waste within the 100-d period of the experiment. In a separate study, food waste was digested using a two-stage process to generate hydrogen gas and CH_4 (Han and Shin, 2004). In this process, 72.5% of the VS was removed. Kayhanian and Tchobanoglous (1992) evaluated the degradability of food waste based on lignin content and determined that 82% of the VS could be decomposed under anaerobic conditions.

Table 3. Methane yield (in L and mol of CH_4), time required for decomposition, and equivalent Mg CO_2 for different types of wastes incubated in a laboratory to simulate decomposition in a municipal solid waste landfill (Eleazer et al., 1997).

	Yield			Time
	L CH_4 kg^{-1}	mol CH_4 kg^{-1}	Mg CO_2 Mg^{-1} waste	Days
Grass	144.4	6.4	2.37	50
Grass-2	127.6	5.7	2.10	50
Leaves	30.6	1.4	0.50	100
Branch	62.6	2.8	1.03	100
Food	300.7	13.4	4.94	120
Coated paper	84.4	3.8	1.39	150
Old newsprint	74.33	3.3	1.22	300
Corrugated containers	152.3	6.8	2.50	400
Office paper	217.3	9.7	3.57	500

The values for CH_4 generation from grasses of 127.6 to 144.4 L CH_4 kg^{-1} (or 0.13–0.14 m^3 CH_4 kg^{-1}) observed in Eleazer et al. (1997) were also lower than values obtained in anaerobic digesters. When based on the VS content of the grasses (87%), the values from Eleazer et al. (1997) correspond to 0.15 to 0.17 m^3 CH_4 kg^{-1} VS. Mähnert et al. (2005) looked at the CH_4 generation potential of three different grass species in a mesophilic anaerobic digester with an HRT of 28 d. Biogas yields ranged between 0.65 and 0.86 m^3 kg^{-1} VS, with between 60 and 70% of the gas consisting of CH_4. This corresponds to 0.31 to 0.36 m^3 CH_4 kg^{-1} VS. The authors quote a range of other studies using different fresh cut grasses or grass silage where biogas production ranged from 0.5 m^3 kg^{-1} VS to 0.81 m^3 kg^{-1} VS. Methane yield in other studies ranged from 0.23 to 0.5 m^3 CH_4 kg^{-1} VS. The low values observed by Eleazer et al. (1997) may be related to the type of grass, the lack of mixing, or the relatively high loading rate used.

Paper waste is the final component of MSW that has the potential to generate significant quantities of CH_4 in a landfill. In the Eleazer et al. (1997) study, this category of MSW was separated into individual components, including newsprint, coated and corrugated paper, and office paper. Methane generation from these individual categories ranged from 74.3 L CH_4 kg^{-1} for old newsprint to 217.3 L CH_4 kg^{-1} for office paper. Both materials have a VS content of 98%. Duration of CH_4 emissions ranged from 120 d for coated paper to 500 d for office paper. Clarkson and Xiao (2000) also looked at the CH_4 generation potential of newsprint and office paper in bench scale anaerobic reactors. They found that CH_4 generation from office paper was largely complete after 20 d but continued until Day 165, with yield ranging from 71 to

Table 4. Methane generation and CO_2 equivalents for food waste that has decomposed in anaerobic digesters or simulated landfills.

Source		Volatile solids destruction	L CH_4 kg^{-1}	mol CH_4 kg^{-1}	Mg CO_2 Mg^{-1} waste	Days
Eleazer et al., 1997	simulated landfill		300.7	13.4	4.94	120
Zhang et al., 2007	anaerobic digestion		348	15.5	5.72	10
		81	435	19.4	7.15	28
Nguyen et al., 2007	anaerobic digestion	61	260	11.6	4.27	60
Cho and Park, 1995	anaerobic digestion		489	21.8	8.03	40

Table 5. Summary of the maximum potential volatile solids (VS) reduction and CH_4 generation capacity of different feedstocks that are suitable for stabilization by composting.

Feedstock	Maximum% destruction VS	CH_4
Manure		
Dairy cow	48	120 g CH_4 kg^{-1} manure
Swine	50	141 g CH_4 kg^{-1}
Chicken	60	179 g CH_4 kg^{-1}
Municipal biosolids	80	0.364 kg CH_4 kg^{-1} VS VS destroyed
Municipal solid waste		
Food waste	70	190 g CH_4 kg^{-1}
Grass	87	101 g CH_4 kg^{-1} VS
Paper waste		
Newsprint	22	47 g CH_4 kg^{-1}
Office paper	82	138 g CH_4 kg^{-1}
Mixed paper	67	134 g CH_4 kg^{-1}

85% of chemical oxygen demand (COD) and total production of about 300 mL CH_4 g^{-1} COD. In comparison, CH_4 conversion from newsprint ranged from 32 to 41% of total COD for 300 d or about 100 mL CH_4 g^{-1} COD. The authors cite previous studies that showed only a 20% conversion of newsprint COD to CH_4. The lignin content of the newsprint and the association of lignin with cellulose are the reasons given for the slow and limited decomposition of this type of paper. Kayhanian and Tchobanoglous (1992) also used the lignin content of different paper streams as a basis for determining the degradability. Degradability was defined as the decomposition under anaerobic conditions for a 30- to 45-d retention time. For office paper, 82% of the VS was considered degradable. Only 22% of the VS from newsprint was degradable due to the high lignin content. A total of 67% of the VS fraction of mixed paper was degradable. The values from these studies are not dissimilar and suggest that the CH_4 generation potential predicted by Kayhanian and Tchobanoglous (1992) offers a simple means to measure the CH_4 generation potential of the paper based on the initial VS content.

Summary

Using the different values from different reports and research studies, it is possible to summarize the CH_4 generation potential for each type of feedstock considered in this review. These values can be used as a baseline for determining if GHG avoidance credits are appropriate for composting operations where feedstocks are composted that otherwise would have been landfilled or stored in anaerobic lagoons without gas capture. Table 5 summarizes the values from the proceeding section. The IPCC has discounted the potential for N_2O from landfills. The small number of studies suggests that as more data become available, this may be revised (Béline et al., 1999; Rinne et al., 2005).

Greenhouse Gas Potential from the Composting Process

The composting process has the potential to accrue GHG debits because of the energy required for the composting process and the unintentional release of GHGs during composting. Different types of composting operations and energy requirements for each are detailed herein. The potential for fugitive GHG release before composting when feedstocks are stockpiled and during the composting process is discussed.

Feedstock Mixing

In some cases, materials to be composted require grinding or sorting before composting. Also, some high-moisture feedstocks (like biosolids and some manures) require a bulking agent, which may also require size reduction. This process requires energy. In a life cycle analysis of composting of yard wastes, a tub grinder running at 1200 HP used 200 L h^{-1} of fuel (Recycled Organics Unit, 2006). During this time, the unit was able to grind 60 Mg of feedstock. A tub grinder on the market in the USA, the Vermeer TG9000, comes in 800- and 1000-HP units that can grind 50 to 100 tons of feedstock per hour (http://www.vermeer.com/vcom/EnvironmentalEquipment/Index.jsp?NewsID = 12752). These examples can be used to develop energy estimates for grinding feedstocks. Grinding requires about 3.3 L of fuel per wet ton. At 50% moisture, this is equivalent to 6.6 L per dry ton. If 1 L of fuel weighs 0.845 kg and 1 kg of fuel is taken to be the equivalent of 3.28 kg CO_2, mixing 1 dry ton of material has a CO_2 emission cost of 18.3 kg (Recycled Organics Unit, 2006). These examples can be used to develop energy estimates for grinding feedstocks. The feedstocks that typically need grinding include woody debris and yard waste. Nutrient-rich materials, including manures and biosolids or materials coming from anaerobic digesters, are much more homogenous in appearance and do not require preparation before composting. However, some high-moisture feedstocks require bulking agents to provide structure and porosity, and other feedstocks (e.g., food scraps) may require mechanical mixing before being composted.

Composting operations can generally be grouped into three major categories. These include windrows, static piles with or without forced aeration, and mechanized processes (Savage et al., 1993; Haug, 1993; IPCC, 2006). Each of these requires different amounts of energy to set up and manage materials while they are composting.

Composting Process Energy Requirements

Windrow Systems

Windrow composting consists of feedstocks mixed at appropriate ratios and set up as long rows that are mechanically turned to accelerate the composting process. The Recycled Organics Unit of the University of New South Wales (Recycled Organics Unit, 2006) conducted a life-cycle analysis for windrow composting of yard wastes that included GHG estimates for the composting process. Their analysis included transport of materials to the compost facility, shredding and blending feedstocks, setting up and turning piles, and transport of the finished product to end users. The estimate was based on a facility that processes 40,000 Mg of garden organics per year. With the windrow system, this was calculated to require 200,000 L of diesel fuel or 5 L Mg^{-1} of feedstock. It was based on the assumptions that initial mixing would be done with a loader, piles would remain on site for 16 wk with turning every 3 to 4 wk, and material would require shredding before mixing.

The estimate was done for a pile that is 1.5 to 3 m high, 3 to 6 m wide, and of indefinite length. Estimates of GHG emissions associated with fuel use included production and combustion emissions (total 3.28× weight of fuel used for CO_2 equivalent). The estimate included electricity for operating office facilities and the compost facility totaling 150,000 kW hours (50,000 of which were for office operations). The electricity used on site was for collecting and pumping rain water to keep piles moist. Sprinklers were operated 10 h d^{-1} for 7 to 10 d during the initial 4 wk of the composting process. This is equivalent to 4 kW per ton of compost. Based on using coal-fired power plants for electricity, each MJ of electricity was equivalent to 0.28 kg CO_2. This equivalence would need to be adjusted on a facility-specific basis if a portion of the energy used at the facility came from green sources. Because this estimate did not include gas emissions during composting, the energy used to produce the compost was the most important source of GHG debits for the composting process.

One of the primary reasons that biosolids are composted is to eliminate pathogens. For composting to be used as a mechanism to destroy pathogens, the USEPA has specified a minimum temperature and number of turns before the material can be called "Class A," or pathogen free (USEPA, 1993). For windrow compost piles, the requirements are for five turns and maintaining a temperature of 55°C for 72 h between each turn. This is within the range of the number of turns outlined in the Recycled Organics Unit analysis and suggests that energy requirements for windrow compost systems to achieve Class A pathogen reduction requirements in the USA would be similar to the projections by the Recycled Organics Unit.

The USEPA (2002) did an analysis of GHG potential from a windrow compost system and concluded that processing 1 Mg of feedstock requires 5.90 kg of diesel fuel (for material processing and turning). Their energy cost estimate is expressed as 0.01 metric tons carbon equivalent of indirect CO_2 emissions per ton of material composted, with the estimate including transport of material to a compost facility (363,000 Btu) and turning the pile (221,000 Btu). This estimate is much less detailed than the life cycle analysis done by the Recycled Organics Unit. It does not include equipment requirements or the number of turns required for composting. However, both estimates for fuel use are similar. Operations with more frequent turning of piles or with piles of larger or smaller dimensions would require some adjustments to this estimate. Based on the detail provided by the Recycled Organics Unit, it should be possible to alter the estimate with data from specific facilities.

Aerated Static Pile Systems

For aerated static pile systems, air needs to be forced through the system to maintain aerobic conditions through the pile and to control temperature. The classic aerated static pile systems is known as the Beltsville static pile because it was developed at the USDA research facility in Beltsville, Maryland (Savage et al., 1993). This system was initially developed for municipal biosolids and therefore is suitable for feedstocks with high moisture content. High-moisture feedstocks are mixed with a bulking agent to provide structure and porosity. The mix is piled on perforated pipes that have been covered with finished compost or the bulking agent. The pile is similar to that described in the section on windrow composting. The pile is then generally covered with a layer of finished compost, which maintains heat within the pile and reduces odor emissions from the pile. Air required to maintain aerobic conditions changes with the specific environmental conditions and with the feedstock. Air is generally turned on for set intervals during the composting period. Systems can be controlled automatically based on temperature. Air requirements for these systems can vary.

Savage et al. (1993) provide a general guide that blowers need to be sufficiently powerful to pull 0.7 to 2.4 m^3 min^{-1} air per dry ton. Haug (1993) modeled a static pile system that consisted of 44 dry Mg of municipal biosolids with a solids content of 25% with wood chips added as a bulking agent. This is equivalent to 176 Mg total weight. His calculations include aeration for a 21-d period with total air required averaging 0.036 m^3 of air per minute per 0.093 m^2. A working guideline is that 0.042 m^3 of air are required to aerate 0.093 m^2 of mixture. Haug gives the ideal moisture content for a compost feedstock containing biosolids at 55 to 60%. This material would have a density of 0.8 g cm^3. If a pile were constructed based on the dimensions recommended by Savage et al. (1993) (24.4 × 4.3 × 2.1 or 224 m^3), the total weight of the pile would be 89 Mg. This is equivalent to 16.8 cfm per US ton for the composting period. Haug (1993) says that although static pile systems initially supplied with 0.24 m^3 min^{-1} ton^{-1} dry solids, aeration rates have increased and are now approximately 0.94 m^3 min^{-1} ton^{-1} solids. Using 0.24 m^3 min^{-1} ton^{-1} as a low range and 0.94 m^3 min^{-1} ton^{-1} as a high range would seem to be a conservative approach. For specific operations, the blower capacity and electric requirements can be calculated using these values as lower and upper boundaries. The amount of energy required can be calculated using a carbon equivalent of 0.66 kg CO_2 kWh^{-1} of electricity used (Amon et al., 2006).

In addition to the energy needed to move air, the piles also require energy to set up and break down. The equipment required to do this is similar to that required to set up a compost windrow because the static pile system is essentially a windrow that is not turned. For this, the fuel estimate prepared by the Recycled Organics Unit can be adapted to fit a static pile system. In that estimate, 5 L of fuel was required to process 1 Mg of feedstock, including mixing and setting up, turning, and breaking down piles. Here mixing, setting up, and breaking down piles consume 2.5 L of fuel for each Mg of feedstock.

Mechanical Systems

Mechanical systems are based on the same concepts as windrow and static pile systems, but a portion of the composting process takes place in an enclosed structure (Haug, 1993). There are a wide range of types of structures that fall into this general category, including vertical flow and horizontal flow systems. In both of these types of systems, the mixed feedstock is introduced into one end of the reactor and is gradually moved through a system until it is sufficiently stabilized and exits the enclosed system. The movement is generally combined with forced aeration. In many cases, in addition to the mixing that occurs as the feedstocks are moved through the system, feedstock mixing is accomplished using augers. The retention time in these systems is often short, so even though some decomposition occurs within the reactor, the material requires additional curing time in windrows before it is stable enough to be marketed.

With the range of different systems available, it is out of the scope of this review to detail energy requirements for each individually. The vast majority of composting systems in the USA are windrow systems, followed distantly by aerated static pile systems, with relatively few proprietary commercial systems in operation. However, it is possible to outline general guidelines for evaluating these requirements that can be used for specific systems. Each of these systems includes some type of aeration. They also involve mechanical moving of feedstocks through the system from the top down or from the point of entry to the point of exit. Finally, they require energy for loading and mixing the substrates and energy for setting up the curing piles that the composts are put into to stabilize after they are through the reactor.

Steve Diddy (personal communication) provided information on energy requirements for a specific enclosed system that his company is recommending. The system can handle up to 137 dry Mg of feedstock per day for a 365-d work year or a total of 50,000 dry Mg yr^{-1}. The energy requirements for aerating this system average 90 kWh per dry Mg. Depending on the source of electricity in the region where the system is used, the energy costs can be high. One source gave a carbon equivalent of 0.66 kg CO_2 kWh^{-1} of electricity (Amon et al., 2006). This is a per ton electricity cost for aeration of approximately 60 kg CO_2 Mg^{-1} of material.

Greenhouse Gas Emissions During the Composting Process

Background

Several prior reviews discount the potential for GHG to evolve during the composting process or during storage of feedstocks before composting (Recycled Organics Unit, 2006; Smith et al., 2001; USEPA 2002). They argue that a well run operation maintains aerobic conditions to maximize the rate of decomposition. There are marked differences in decomposition rates in an aerated versus an anaerobic system. In one study, turning a pile four times during a 4-mo composting period resulted in a 57 ± 3% reduction of initial organic matter. The same feedstocks maintained in a static pile for the same period had a 40 ± 5% reduction of initial organic matter (Szanto et al., 2007). In addition, the Recycled Organics Unit (2006) justifies this because of the low moisture content and low bulk density of yard wastes. Although anaerobic conditions may exist in a pile, high ammonia concentrations within the pile are sufficient to inhibit methanogenic bacteria. An additional safeguard against the release of any GHGs formed within a pile is the active aerobic microbial community on the surface of the pile. These organisms oxidize CH_4 before it is released into the environment (Bogner et al., 1997).

In two of these estimates, the focus was on yard waste or municipal solid waste including yard wastes for feedstocks (USEPA, 2006; Recycled Organics Unit, 2006). Another study of gas emissions during composting when using yard waste also found low GHG emissions. In a lab incubation using yard waste as a feedstock, 12 to 114 g N_2O–N per Mg of feedstock was emitted over an 89-d period (Hellebrand and Kalk, 2001). Yard wastes typically have a lower moisture content, lower bulk density, and a higher C/N ratio than feedstocks such as animal manures and municipal biosolids. In the case of yard waste composting in Australia, moisture is commonly limiting. This is not the case for all compost facilities or for different feedstocks. In situations where wet feedstocks with low C/N ratios are used, there is a potential for CH_4 and N_2O emissions from compost piles. There is also the potential for GHGs to volatilize from feedstocks because they are stored before composting. For storage and composting, there may be operational factors that can be modified to reduce the potential for GHG evolution. Literature on GHG evolution from feedstock storage and during composting is summarized below.

Other Feedstocks—Storage

Although feedstocks such as grass clippings and food waste are generally landfilled almost immediately after they are generated, animal manures are often stored before they are used. There have been a number of studies to quantify CH_4 and N_2O emissions from animal manures. These studies have included GHG generation of materials that have undergone different types of storage and treatment before storage. Results from these studies indicate that storage of manures can be a significant source of GHGs and that management practices can be used to reduce these gases.

Béline et al. (1999) altered the retention time and oxygen flow for aerobically digested pig manure. They observed N_2O formation for all treatments and attributed this to nitrification and denitrification reactions. According to the authors, these reactions can occur when oxygen is between 1 and 10% of saturation or conditions are moderately reducing. Although this study was done during treatment rather than storage, it illustrates the importance of microenvironments for the generation of N_2O. The importance of low oxygen levels in the formation of N_2O has been recognized by other authors (Wrange et al., 2001). The C/N ratio of the feedstock can also be an important variable. In feedstocks with a high C/N ratio, N is limiting and is more likely to be immobilized than denitrified. Yamulki (2006) added straw to manure heaps and found that CH_4 and N_2O emissions were reduced when the C/N ratio of the mixture was increased. Reductions as a % of loss of N and C as N_2O and CH_4 were from 0.7 and 0.06 to 0.48 and 0.03 for N_2O and CH_4, respectively. This was attributed to the reduction in available N and to the lower moisture content of the mixture. Increased porosity leading to higher O_2 diffusion may also have been a factor. Although Béline et al. (1999) did not report the C/N ratio of the pig manure, the high N content (3–4 g kg^{-1}) suggests a low C/N ratio. They also tested N_2O formation in manure that had been stored anaerobically in an enclosed space after aerobic digestion. In the last case, N_2O was formed and subsequently transformed into N_2 during storage.

Amon et al. (2006) looked at N_2O and CH_4 from cattle manure from a dairy operation treated in a variety of ways and then land applied. The C/N ratio of the slurry was 8:1. Treatments included tank storage, separation of liquid and solid phases followed by composting of solid and land application of liquid, anaerobic and aerobic digestion, and storage with a 10-cm straw cap. Gas emissions during storage and land application were measured. For all options, over 99% of the total CH_4 emissions occurred during storage of the slurry. The straw cap did not reduce CH_4 emissions from the stored slurry. The majority of N_2O (77.6–91.3%) was also released during storage. Clemens et al. (2006) looked at the effect of temperature and storage cover on gas emissions from cattle slurry (Table 6). Raw and digested manure were tested. Methane emissions increased significantly with warmer temperatures. Prior anaerobic digestion in combination with a cover decreased CH_4 production. In another study, Sommer et al. (2004) modeled CH_4 and N_2O release from pig and cattle slurry in house and after field application. They used in-house storage for 1 yr with field applications of the slurries in April in Denmark as the basis for their model. Methane emissions were reduced by increased frequency of washing of manure to outdoor storage areas where cooler temperatures slowed anaerobic decomposition. Nitrous oxide release was modeled only for field application due to the lack of data for stored slurries.

Research on gas emissions from stockpiles of other types of feedstocks is limited. Greenhouse gas generation potential for stockpiled municipal biosolids is expected to be similar to values observed for anaerobically digested manure because the moisture content and C/N ratios are similar. Hansen et al. (2006) measured CH_4 production from anaerobically digested municipal wastes. This study was modeled after practices in Denmark where wastes are kept in digesters for 15 d and then stored for up to 11 mo before a very limited land application window. Methane generation was measured at several temperatures to mimic conditions in outdoor storage tanks at different times of the year. Methane production increased significantly starting at 22°C with a retention time of 120 d (>43 m^3 CH_4 Mg^{-1} VS) and peaked at 55°C (>215 m^3 CH_4 Mg^{-1} VS). This is equivalent to 27 kg CH_4 at 22°C and 136 kg CH_4 at 55°C Mg^{-1} VS. These results are similar to those observed for animal slurries in that increasing temperature is a major factor in determining the quantity of gas that is released from stockpiled feedstocks.

Based on the results of these studies, there are several best management practices that can be adopted to reduce the potential for GHG generation from stockpiled feedstocks before composting. As the rate of gas production increases with increasing temperature, reducing storage time during summer months or in warmer climates limits gas release. If material is stored, it should be stored in a covered facility. If material is wet and has a low C/N ratio, a dry carbonaceous feedstock should be incorporated into the material to increase the final C/N ratio to >20:1 to dry the material and reduce the potential for denitrification. If these practices are not adopted, stockpiling of feedstocks can result in emissions of CH_4 and N_2O.

Composting Process

For feedstocks that are sufficiently nutrient rich and wet, there are a number of studies in the literature that document GHG release from compost piles. A summary of several studies is presented below and in Table 7. It is possible to make generalized conclusions from these studies. The summaries below are followed by a more specific discussion on the relationship between different variables and GHG emissions.

Table 6. Methane and N_2O release from digested and raw cattle slurry. The importance of temperature and management options is shown. Data are from Clemens et al. (2006).

	Winter		Summer	
	CH_4	N_2O	CH_4	N_2O
	g m^{-3}			
Raw-crust	164	44	3590	49
Raw-cover	142	38	3000	57
Digested	111	40	1154	72
Digested-straw	115	40	1192	76
Digested-straw-cover	81	41	1021	61

Hao et al. (2004) looked at emissions from cattle feedlot manure composting with straw and wood waste as bulking agents. Manure was set in windrows that were turned eight times over a 100-d composting period. Air samples were collected from the top of the windrows in the morning several times per week during the initial stages of the composting operation and then on weekly intervals. Methane was detected in both windrows during the first 60 d of the process, with over 50% being released by Day 30. These results were similar to those observed in other studies (Fukumoto et al., 2003; Lopez-Real and Baptista, 1996; Sommer and Moller, 2000). This makes sense because over time the piles would dry out and become more consistently aerobic. In the Hao et al. (2004) study, N_2O was detected in the early stage of the process and again in the last 30 d of the process. The authors suggest that this was the result of denitrification of NO_3^-. Total emissions for each bulking agent were similar and equaled approximately 8.92 kg C as CH_4 Mg^{-1} dry weight with 0.08 kg N Mg^{-1} as N_2O.

In another study, Hellebrand and Kalk (2001) observed CH_4 (1.3 kg m^2) and N_2O (12.8 g m^2) emissions from windrow composting of animal manures on an organic farm. As in Hao et al. (2004), CH_4 evolution was limited to the initial composting period. Nitrous oxide emissions peaked during the third week of composting. Data were not given on the gas evolution on a dry-weight basis. Analysis of feedstocks and moisture content of the piles are not reported. Fukumoto et al. (2003) composted swine manure in a large and small conical pile to investigate the role of pile size in quantity of GHG emissions. Manure was mixed with sawdust; however, the C/N ratio of the final mixture is not given. Total moisture content of both piles was 65%. Methane and N_2O emissions increased with increased pile size. The authors attribute this increase to the increased anaerobic sites within the larger pile. They attribute the formation of N_2O to nitrate from the aerobic portion of the pile being mixed into the anaerobic section. A total of 37.2 and 46.5 g N_2O kg^{-1} N and 1 and 1.9 g CH_4 kg^{-1} OM were formed from the small and large piles, respectively. Hellman et al. (1997) observed CH_4 and N_2O release from the surface of co-composting MSW and yard waste. The moisture content of the initial pile was 60%, and the C/N ratio was 26:1. Gases were collected from the top of the pile at a fixed time of day for

Table 7. A summary of research reporting N_2O and CH_4 emissions from composting operations. The citation, feedstocks used, type of compost system, moisture content, and quantity of gas evolved are shown.

Reference	Feedstock	System	% Moisture	CH_4 loss	N_2O loss
Hao et al., 2004	cattle feedlot manure + straw	windrow	60%	8.92 kg C per Mg manure 2.5% of initial C	0.077 kg N Mg manure 0.38% of initial N
	cattle feedlot manure + wood chips	windrow	60%	8.93 kg C Mg 1.9% of initial C	0.084 kg N Mg manure 0.6% of initial N
Hao et al., 2001	cattle manure and straw bedding	static pile	70%	6.3 kg CH_4–C Mg^{-1} manure	0.11 kg N_2O–N Mg^{-1} manure
		windrow	70%	8.1 kg CH_4–C Mg^{-1} manure	0.19 kg N_2O–N Mg^{-1} manure
Fukumoto et al., 2003	swine manure + sawdust	static pile– no aeration	68%	1.9 kg Mg^{-1} OM (0.5% of initial C)	46.5 kg N Mg^{-1} N 4.6% of initial N
Lopez-Real and Bapatista, 1996	cattle manure + straw	windrow	75%	background	not measured
		aerated static pile	75%	background	
		static pile	75%	48,675 ppm per volume	
Sommer and Moller, 2000	pig litter, low straw	static pile	76%	191.6 g C 0.2% of initial C	58.6 g N 0.8% of initial N
	pig litter, high straw		35%	BD†	BD
Hellebrand and Kalk, 2001	cattle, pig manures + straw	windrow		1.3 kg m^{-2}	12.8 g m^{-2}
Hellman et al., 1997	yard waste + MSW	windrow	60%	252 g C–CH_4	54 g N–N_2O
He et al., 2001	food waste	aerated static pile	65%	not measured	4 μL L^{-1} for 60 d
Czepiel et al., 1996	biosolids + wood ash	aerated static pile	75%	not measured	0.5 kg N_2O Mg^{-1} dry feedstock (1.3% of initial N)
	manure + seasoned hay	windrows	not reported	not measured	0.125 kg N_2O Mg^{-1} dry feedstock
Beck-Friis et al. 2001	food waste	aerated static pile	65%	not measured	<0.7% of initial N
Kuroda et al. 1996	swine manure + cardboard	windrow	65%	very low	0.1% of initial N

† BD, below detection.

each sampling event. Methane release ranged between 0.5 and 1.5 g C h^{-1} Mg^{-1} dry weight from Day 10 to Day 28, after which time it fell below detection limits. Nitrous oxide emissions began after Day 30 and ranged from <50 mg N_2O–N h^{-1} to >100 mg N_2O–N h^{-1} for the next 30 d. Szanto et al. (2007) composted pig manure mixed with straw in astatic pile and a turned pile. The initial C/N ratios of the feedstocks ranged from 14 to 7, and moisture content of the piles was approximately 70%. Turning resulted in no emissions of CH_4 for the duration of the composting period and resulted in lower emissions of N_2O.

Czepiel et al. (1996) measured N_2O emissions from composting municipal biosolids and livestock waste. The biosolids were composted in an aerated static pile with wood ash used as a bulking agent. The manure was composted in windrows that consisted of 25% manure and 75% seasoned hay from bedding. Only the moisture content of the biosolids is given (75%). Nitrous oxide and O_2 content as well as the fraction of the pore space occupied by water are given for the biosolids compost at Day 9 and Day 38. As with other studies, N_2O concentration increased with limited, but not depleted, O_2. Total N_2O generation capacity for the manure compost pile was given as 0.5 g N_2O–N kg of manure or 0.125 g N_2O–N per dry kg feedstock. Total N_2O generation capacity for the biosolids compost pile was given as 0.7 g N_2O per dry kg biosolids. The amount of ash included in the mix was not provided. In a lab study using household waste with a C/N ratio of 22:1 and total C content of 38%, Beck-Friis et al. (2001) reported that of the 24 to 33% of initial N that was lost during the composting process, <2% evolved from the system as N_2O–N. This is equivalent to 0.1 g N as N_2O for each kg of feedstock or 100 g Mg^{-1} of feedstock. Moisture content, C/N ratio, and aeration status seem to be key variables in controlling GHG emissions from compost piles, with moisture content perhaps the most important.

Moisture Content

In a general discussion of composting, Haug (1993) discusses appropriate moisture contents for different feedstocks and points out that when a feedstock is too wet, it is difficult to maintain an aerobic environment that is conducive to rapid composting (Table 8). Ideally, a compost pile should not be oxygen or moisture limited. Feedstocks that are able to maintain some structural strength including straw or wood chips can have higher % moisture than others that will subside and eliminate pore spaces. The latter category of materials includes manures, biosolids, and wet wastes such as food waste and grass clippings. Methane, in addition to being a GHG, is indicative of anaerobic conditions that are associated with slower decomposition or composting in comparison to aerobic conditions.

In all of the studies where GHG emissions have been detected from the surface of the compost piles, the % moisture of the feedstocks has been at or above the maximum levels for appropriate aeration. The importance of moisture content in controlling GHG emissions is made clear by the following study. Sommer and Moller (2000) looked at the effect of varying straw content on GHG emissions from pig litter. Two rates of straw addition were used, one to bring the bulk density of the mixture to 0.23 g cm^3 and the other to to bring the bulk density to 0.44 g cm^3. The dry matter content and C/N ratio of the piles also differed: 23.6% (DM) and 12.76 (C/N) for the low-straw and 65.2% and 16.27 for the high-straw piles. No CH_4 and minimal N_2O (0.1–0.5 g N m^2 min^{-1} for approximately 3 d at the beginning of the composting period) were emitted from the dryer pile. The denser pile had significantly higher emissions of both gases. The authors attribute this difference to higher O_2 content measured in the less dense pile. In addition, the convection of hot air from the interior of the pile drove off ammonia, which restricted formation of N_2O in the less dense pile. The results of this study were mirrored in a comparative study of three composting systems for cattle manure (Lopez-Real and Baptista, 1996). Here a mixture of cattle manure (≅70%) and straw were composted using three systems: windrow, forced aeration, and static pile with no aeration or turning. The moisture content of the feedstocks was about 75% for all piles. Although CH_4 concentrations in the piles where air was incorporated were similar to background concentrations, those from the wet, static pile were three orders of magnitude higher for several weeks.

The IPCC has published default values for CH_4 and N_2O emissions for different types of composting operations using swine and cattle manures as feedstocks (IPCC, 2006). Different parameters, including moisture content and C/N ratio, are not included in these estimates. The CH_4 estimates were based on the best judgment of the authors and include a reference to a single published study. Methane conversion factor estimates range from 0.5% for in-vessel composting under all temperature regimes to 1.5% for passive and intensive windrow composting operations. Estimates for N_2O emissions from composting animal manures range from 0.006 to 0.1 kg N_2O–N kg^{-1} N excreted for static pile and intensive windrow systems, respectively. These estimates are based on the judgment of the IPCC group rather than a review of the existing literature.

Pile Surface Reactions

The previous section detailed the importance of feedstock variables, including C/N ratio and % solids, in controlling fugitive GHG emissions from compost operations. Emissions of GHGs can be controlled by the active microbial community on the surface of the piles that oxidize (CH_4) or reduce (N_2O) gases before they are emitted (USEPA, 2002). There is evidence for this in the literature. Hao et al. (2001) measured gas concentrations at a range of depths from windrowed manure compost piles. One of the piles was turned, and the other was left to cure without turning. The feedstock used for both piles was relatively wet, with a solids content of 30%. Both CH_4 and N_2O were detected in the piles during the 90 d that the piles were monitored, CH_4 consistently and N_2O periodically over the course of the composting period. Concentrations of both gases were highest below the pile surface. The highest CH_4 concentrations (<15% of gas collected) in the turned pile were found at the bottom of the windrow with concentrations in the top 46 cm ranging from 0 to 3% for the entire composting period. Concentrations of N_2O were detectible in the turned pile only at T71d and T84–90d. The highest measured concentration was between 20 and 550 μL L^{-1}. However, this concentration was only observed at the 30- to 107-cm depth. Concentrations in the upper portion of the pile were at the detection limits. This pattern was also observed in the static pile with higher concentrations of both gases throughout the composting period.

Kuroda et al. (1996) measured emissions of odorous compounds and GHG from windrowed swine manure mixed with corrugated cardboard to bring the moisture content to 65%. Methane was only detected in the first day of the composting process. Nitrous oxides and odorous compounds (methyl mercaptan, dimethyl sulfide, and dimethyl disulfide) were detected in gradually decreasing concentrations through the first 28 d of the composting process. For each of these (less pronounced for dimethyl disulfide), emissions peaked immediately after the pile was turned and then sharply decreased until the next turn. This pattern suggests that the gases were present in the interior of the piles and were released as the material from the interior was brought to the surface. When not being turned, diffusion of these gases to the atmosphere through the pile was limited by decomposition of the compounds in the aerated surface section of the pile. Total release of N_2O in this study was 0.1% of total initial N. This figure is consistent with other research. In a cattle manure/straw static pile system, Lopez-Real and Baptista (1996) observed CH_4 concentrations in the pile of up to 48,675 mg kg^{-1} per volume, whereas concentrations on the surface of the pile were 2620 mg kg^{-1} per volume.

Table 8. Maximum recommended moisture content for various composting materials (from Haug, 1993).

Type of waste	Moisture content
	% of total weight
Straw	75–85
Wood (sawdust, small chips)	75–90
Rice hulls	75–85
Municipal solid waste	55–65
Manures	55–65
Digested or raw sludge	55–60
Wet wastes (grass clippings, food waste etc.)	50–55

Other researchers have used finished compost as a means to oxidize CH_4 from landfills before it is released to the atmosphere. Compost has also been used to construct biofilters to oxidize CH_4 (Mor et al., 2006). Thermophilic methanotrophs have been identified in compost piles that oxidize CH_4 before it is released to the atmosphere (Jäckel et al., 2005). Finished compost, as a surface to active compost windrows, has also been used to reduce VOC emissions from the windrows (Fatih Buyuksonmez, personal communication). These data suggest that the microbial community in a compost pile is active enough to prevent or reduce the release of GHGs from the interior of the pile to the atmosphere. It also suggests that controls to reduce odor emissions from composting piles, including biofilters and capping piles with finished compost, can also reduce GHG emissions from piles.

From the data presented in this discussion, there seem to be particular characteristics of compost feedstocks and composting operations that can be used to determine the potential for GHG release during the composting process.

- If feedstocks are low in nutrient content (C/N >30:1) and/or moisture content (% moisture <55%), then the potential for GHG release during composting can be discounted. Materials such as yard waste, certain agricultural wastes, and mixed MSW can be included in this category.
- If feedstocks include nutrient-rich and wet materials (including animal manures, municipal biosolids, food wastes, and grass clippings), there is a potential that they will release GHGs when they are composting.

- If a bulking agent is added to bring the moisture content to <55% and/or the C/N ratio to >30:1 and/or some type of aeration system is included as a part of the composting process (windrow or aerated static pile), this potential can be discounted.
- If this is not the case, a debit can be taken in relation to the total C and N concentrations of the feedstock. A conservative value for this is 2.5% of initial C and 1.5% of initial N. These values are in agreement with the upper-end values provided by the IPCC (2006).
- If the facility has an odor-control mechanism in place, including scrubbers that oxidize reduced sulfur compounds or a biofilter, debits for CH_4 can be eliminated. If it is a static pile system and the composting feedstocks are covered by a layer of finished compost that is kept moist, the gas emission potential can be cut by 50% for CH_4.

End Use of Composts

The benefits associated with compost use in relation to GHG emissions were considered by the USEPA (2002), Recycled Organics Unit (Recycled Organics Unit, 2006), and Smith et al. (2001) in their calculations. The EPA estimate and Recycled Organics Unit based their calculations on specific end-use cases. Smith et al. (2001) based their calculations on more general properties of compost and the potential for compost to replace peat for a range of end uses.

For their estimate, the Recycled Organics Unit modeled two types of end use for compost: (i) as a soil conditioner for cotton with an application rate of 12 Mg ha^{-1} and (ii) as a mulch for grapes with an application of 75 Mg ha^{-1} every 3 yr. Factors that were considered included increased soil C, reduced water usage, fertilizer value, and reduced use of herbicides. The Australian regulations have product quality criteria for mulch and soil conditioners, with nutrient concentration being a primary determinant. The soil conditioner is a nutrient-rich product with total N of 1 to 2% and a water-holding capacity of 50 to 60%. It contains 55 to 75% organic matter. As a soil conditioner, the potential C credits or benefits associated with compost use were:

- Increased soil water-holding capacity of 2.4 to 3%, resulting in reduced irrigation of 0.13 to 0.16 mL H_2O ha^{-1} in irrigated cotton (reduced energy requirements for irrigation)
- Fertilizer equivalent of 34 to 68 kg N, 29 to 57 kg P, and 24 to 48 kg K ha^{-1} for the first year (reduced energy from avoidance of synthetic fertilizers)
- Sequestering 2.9 to 5.9 Mg C ha^{-1} after 10 yr

Mulch is categorized as a low-nutrient mixture made from garden wastes with 75 to 95% organic matter and total N of 0.2 to 0.4%. The water-holding capacity of the mulch is 10 to 20%. The benefits associated with use of compost as a mulch were detailed as follows:

- Increased soil water-holding capacity of soils by 9.8%, with total savings of 0.95 mL H_2O ha^{-1} for irrigated viticulture
- Fertilizer equivalent of 27 to 40 kg N, 46 to 68 kg P, and 72 to 108 kg K ha^{-1} for the first year
- Herbicide replacement of 2 to 6 L ha^{-1}
- Carbon sequestration of 11.6 Mg C ha^{-1} after 10 yr

Other studies have also shown the ability of compost to increase water-holding capacity and total soil organic matter. For example, Aggelides and Londra (2001) saw improvement in water-holding capacity and other soil physical properties after application of 75, 150, and 300 m^3 ha^{-1} of a finished compost. Improvements were proportional to application rate. Albiach et al. (2001) noted increases in soil organic matter after application of 98 to 122 Mg ha^{-1} of compost. Soil type and local climactic conditions also influence the extent of benefits after compost application.

The USEPA (2002) discussion considered application of 1.3 to 3.2 wet tons (2.9–7.2 Mg ha^{-1}) of compost to field corn. Although this estimate included some discussion of the fertilizer value of compost and changes in soil properties associated with compost use, these variables were not included in any type of quantitative estimate. As a result of this and of the low application rates, benefits from land application of compost were minimal. The different conclusions reached by scenarios modeled by the Recycled Organics Unit and the USEPA illustrate the importance of the type of end use for compost in quantifying the benefits associated with that end use. These differing conclusions suggest that quantifying benefits with regard to GHG and end use of compost will be possible only on a case-by-case basis.

Smith et al. (2001) used a different approach in an attempt to be less case specific. The authors attempted to identify the portion of the C in the compost that would persist for >100 yr. They note that C dynamics vary by soil type and climate and attempt to identify the fraction of the added organic matter that is likely to persist under a range of scenarios. They also consider likely climate changes in Europe as a result of global warming and their effect on soil C dynamics in their estimate. The fraction that will persist is taken to be 8.2% of the added C or 22 kg CO_2 per wet Mg of feedstock. This estimate does not consider the potential for added C to restore soil tilth, which would subsequently allow the soil system to maintain a higher level of C based on increased annual inputs related to improved plant vigor and soil microfauna.

Smith et al. (2001) also consider direct replacement of peat by compost as an additional source of C credits for use of compost. Here they calculate the quantity of peat used by home gardeners and factor the portion of that that could be replaced by compost. For this estimate they consider only the C content of the peat and do not factor in emissions when peat bogs are harvested or when peat is transported to end-use sites. Credits for this end use are 29 kg CO_2 per Mg of feedstock. This type of credit may be valid for compost that is used as a peat replacement. The small quantity of credits associated with this end use suggests that actual accounting to gain these credits may be limited.

Conclusions

This estimate has focused primarily on GHG avoidance for compost feedstocks where compost is chosen as a recycling option rather than landfilling or lagoon storage. Against this, the potential release of GHGs during the composting process were calculated. These include emissions from stockpiles of feedstocks before active composting, energy costs for different compost systems, and GHG emissions from actively composting feedstocks. At this point in the process, the cured composts are ready to be marketed. Although the USEPA and the Recycled Organics Unit estimates included the potential credits associated with compost use, it is not clear if these benefits are appropriate to consider in this estimate. The very different scenarios used by each group resulted in different end credits. The way that both groups went about calculating these benefits were centered on different end uses rather than inherent properties of the materials. Neither estimate considered landscaping uses of compost, which are common near urban centers that produce much of the feedstocks (MSW, food waste, and municipal biosolids) that are commonly used for composting. End use of compost is difficult to monitor; therefore, it may be more appropriate to allocate any GHG credits to the end user who is able to document how the material was applied and the resulting savings. This is also the approach taken by the CDM (2006).

The majority of C credits for composting different feedstocks are related to avoidance or prevention of CH_4 release. Composting operations can also be considered to be sources of GHGs for the energy requirements for composting and for fugitive gas emissions during the composting process. Even in a worst-case scenario, however, these emissions are minimal in comparison to the benefits associated with avoidance credits. It is also possible to significantly reduce emissions from compost piles by an increase in the solids content of the feedstocks and by an increase in the C/N ratio. This review confirms that composting, in addition to other environmental benefits associated with this practice, can be an effective means to reduce GHG emissions from a range of waste materials.

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