ISSUE PAPER

METHANE AVOIDANCE FROM COMPOSTING

An Issue Paper for the:
Climate Action Reserve

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From Science to Solutions

CLIMATE ACTION RESERVE

Table of Contents

List of Tables

List of Figures

Abbreviations

AACP	Alberta Aerobic Composting Protocol
BMP	Biological Methane Potential
C	Carbon
C	Centigrade
CCX	Chicago Climate Exchange
CDM	Clean Development Mechanism
CH_4	Methane
CIWMB	California Integrated Waste Management Board
CO_2	Carbon Dioxide
CTI	Composting Systems International
DOC	Degradable Organic Carbon
DOC_f	Fraction Degradable Organic Carbon dissimilated
g	Grams
GHG	Greenhouse Gas
H_2O	Water
ha	Hectare
IPCC	Intergovernmental Panel on Climate Change
kg	Kilogram
m	meters
M	Mechanical Treatment
MB	Mechanical- Biological Treatment
MCF	Methane Correction Factor
mg	Milligram
Mj	Mega-joules
MSW	Municipal Solid Waste
MT	Million tons
N	Nitrogen
N_2O	Nitrous oxide
NH_3	Ammonia
NO_2-	nitrite
O&M	Operation and Maintenance
OWD	Organic Waste Diversion
PFRP	Process to Further Reduce Pathogens
RDF	Refuse Derived Fuel
SJVAPCD	San Joaquin Valley Air Pollution Control District
SWDS	Solid Waste Disposal Site
t	ton
tCO_2e	Tons Carbon Dioxide Equivalent

UNFCCC	The United Nations Framework Convention on Climate Change
USDA	United States Department of Agriculture
USEPA	United States Environmental Protection Agency
VOC	Volatile Organic Carbon
VS	Volatile Solids

1.0 BACKGROUND

The objective of this issue paper is to reflect and summarize existing research, data, and quantification methodologies related to:

- diverting organic waste from a landfill to a compost facility where it degrades aerobically rather than anaerobically, thus reducing or eliminating methane emissions.

This paper may be used to inform public stakeholder discussions in the development of an actual protocol for quantifying and crediting emission reductions.

This section outlines an introduction to the greenhouse gas (GHG) emissions associated with organic waste and provides an introduction to composting facility activities and potential composting feedstocks.

1.1. Relationship to OWD Protocol

The Organic Waste Diversion (OWD) Protocol is being developed by the Climate Action Reserve and is currently in public draft format. The protocol specifically addresses the following:

> *A biogas control system is designed to capture and destroy methane gas produced from the anaerobic decomposition of organic wastes and manure. By diverting organic waste and manure away from landfills and anaerobic liquid based management systems to a biogas control system, the digestion project is avoiding methane emissions to the atmosphere.*

The OWD protocol also identifies that while composting may be associated with the development of a biogas control system, its does not meet the protocol's definition of a GHG reduction project.

This issue paper relates to the OWD protocol whereby composting facilities also divert organic waste and manure away from landfills and anaerobic liquid based management systems, so providing a similar "baseline" scenario. However, in the case of composting facilities organic waste is degraded in environments that can be anaerobic and aerobic, therefore uncontrolled greenhouse gases can be produced from the "project" case. This provides a unique challenge for identifying GHG reductions from such a project.

It should be noted, that similarly to the OWD protocol, that while the application of composted materials on agricultural soils has the potential to result in substantial GHG benefits, it would be considered a separate GHG reduction activity to a composting facility, which is the topic of this paper.

1.2. GHG Emissions from Organic Waste

1.2.1. Methane

Methane is the by- product of microbial respiration reactions that occur in the absence of oxygen (McMahon et al., 2001). Methane is formed in a multi stage reaction. In the initial stage, volatile solids (including fats, carbohydrates, proteins and complex polymers) are broken down into volatile fatty acids, amino acids, sugars and alcohols by hydrolyzing bacteria. This phase of the reaction is referred to as the acid-forming stage and is carried out by bacteria that are called acid formers. This class of bacteria is relatively pH insensitive and grows rapidly. The second phase of the reaction involves the transformation of these compounds into acetate, hydrogen and CO_2 and is carried out by fermenting bacteria. In the final stage of the reaction the acetate is converted into CH_4 and CO_2. This stage of the reaction is carried out by methane forming bacteria. These bacteria can be divided into two groups: the H2 users and the acetoclastic methanogens. The acetoclastic methanogens consist of two genre, Methanosaeta and Methanosarcina. The chemical transformations are as follows:

$$4H_2 + CO_2 \rightarrow 2H_2O + CH_4$$
$$CH_3COOH \rightarrow CH_4 + CO_2$$

The majority of reactions involve transformation of acetate to CH_4 and CO_2. The methanogens are generally much slower growing than the acid formers and are also sensitive to changes in pH. They require a pH between 6.8 and 7.4 to grow and cannot function in the presence of oxygen gas.

1.2.2. Nitrous Oxide

Nitrous oxide (N_2O) can be formed as an unintentional byproduct of mineralization and nitrification or denitrification reactions. Mineralization and nitrification refer to the microbial processes where organic nitrogen (nitrogen complexed with carbon such as proteins) is transformed into NH_4 and then NO_3-. During nitrification NH_4 is oxidized to nitrite (NO_2-). This is an aerobic reaction that is carried out in soils by bacteria including the Nitrosomonas species. Nitrobacter than oxidizes the NO_2- to NO_3-. Nitrous oxide can be released as an unintentional by-product of nitrification reactions (Bremner and Blackmer, 1978). Nitrous oxide can also be formed by methanotrophic bacteria (Zhang et al., 2009). The bacteria partially oxidize ammonia and release N_2O.

Denitrification describes the process where nitrate is reduced to nitrogen gas. This process is thought to be the primary source of N_2O from organics (e.g. Calderon et al., 2004; Fine et al., 1989, Scott et al., 2000). This is also a microbially mediated reaction. In this reaction the nitrate is used as an electron acceptor in respiration reactions. In aerobic soils or systems, when carbon is used as an energy source, the electrons removed from the carbon are transferred to oxygen. Oxygen gas is converted to reduced oxygen and the oxygen bonds with hydrogen to form water. The equation below describes this general reaction

$$C_6H_{12}O_6 + 6O_2 \rightarrow 6CO_2 + 6\,H_2O$$

In the absence of oxygen, a range of other elements can be used as electron acceptors. These elements include nitrogen, iron, manganese, sulfur and carbon. In each case, progressively less net energy is produced and so the reactions become less and less energetically efficient. When carbon is used as an electron acceptor, the end products are CH_4 and CO_2. Respiration reactions involving nitrogen as an electron acceptor transform NO_3- to N_2. These are very common reactions in wetlands. Wetlands are important environmentally as they are able to remove excess nitrogen from surface waters by converting it into nitrogen gas. Nitrous oxide is an unintentional by-product of these denitrification reactions. A general equation for this reaction is shown below.

$$2NO_3^- \rightarrow 2NO_2^- \rightarrow 2NO\uparrow \rightarrow N_2O\uparrow \rightarrow N_2\uparrow$$

In addition to occurring in wetlands, denitrification can occur in environments where sufficient oxygen isn't available for aerobic respiration and where sufficient NO_3- concentrations are present. For example, Fine et al. (1989) observed N_2O release in soils amended with municipal biosolids only when NO_3- concentrations had increased and only in a high clay, poorly drained soil. In a study done on agricultural soils in Canada, Rochette et al (2008) measured N_2O emissions ranging from 12-45 kg N_2O/ha over the three year period of the study in a high clay soil. Emissions from a neighboring loam soil with identical management practices ranged from 1.0-1.1 kg N_2O/ha over the same period.

There are several factors that are necessary to create an environment favorable to N_2O formation. Oxygen concentration has to be depleted. Sufficient carbon has to be present to provide an energy source for a microbial population. A sufficient concentration of anaerobic soil microorganisms is needed. A high enough concentration of NO_3- must be available for the N to be used as an electron acceptor. If these conditions are met, it is highly likely that some amounts of N_2O will evolve. If anoxic conditions are missing (soil is excessively well drained or coarse), even if a product has a high potential to emit N_2O, emissions are unlikely to be detected (Chantigny, pers. comm.).

1.3. Composting Facility Types and Methods

Commercial and municipal composting facilities in the US use a wide array of technologies from the relatively simple to the mechanically complex. Composting is a biologically mediated process and the fundamental biological principles are the same regardless of scale or technology. Thus to some extent if the biological principles are attended to it can become largely an exercise in materials handling. Numerous developments have been made to try to increase process control or to accelerate process time, while maximizing efficiency in materials handling. The major classes of facilities are discussed below, however there are various iterations within and between the major classes of facilities. These are summarized below and in Table 1. Composting is predominantly an aerobic process. In order to achieve sufficient aeration within a composting system, wetter, denser feedstocks are generally mixed with drier materials that have some structural stability. This allows for airflow and allows aerobic conditions to be maintained. Anaerobic conditions result in slower decomposition, lower temperatures, and malodors.

1.3.1. Passive Piles

Passive piles rely predominantly on natural convection, a function of the porosity (or free air space) of the material or mix being composted. Some passive piles can be very large and rely on anaerobic decomposition as well as aerobic decomposition. Passive piles are often turned very infrequently and may not be suitable for all feedstocks. However, passive piles can have very low odors (as long as the pile is not disturbed). This is the predominant method of mortality composting. Passive piles are likely to contain anaerobic pockets. Temperatures in these piles may not heat up to regulatory requirements for pathogen destruction, one of the hallmarks of commercial and municipal composting.

1.3.2. Turned Windrow

The turned windrow methodology is the predominant method of composting in the US. Generically, windrow composting involves making elongated trapezoidal piles, which are turned with either a tractor, front-end loader, or specialized turning equipment. There can be significant variation in windrow size (which typically depends on the equipment used to turn the pile), windrow length, and management intensity. Some aggressive facility managers can make quality compost in 8 to 9 weeks in an intensively managed windrow operation. Other operators take longer from start to finish. Windrow operations can easily reach temperatures required for pathogen destruction. In these systems, any anaerobic regions are concentrated at the bottom of the windrow. Heat from decomposition effectively dries these piles and in many cases, additional water is added to maintain sufficient moisture for microbial decomposition

1.3.3. Aerated Static Piles

The idea of mechanically forcing air into or through a composting pile was formally researched by the US Public Health Service (now the USDA) in Beltsville, Maryland in the 1970s. Since then forced aeration has been adapted to a variety of feedstocks and a myriad of composting applications. Two important distinctions within aerated static pile systems are whether the air is blown through (positive) or drawn out (negative) of the piles. Other variations include whether the pile is covered with a membrane, whether or not the exhaust air is filtered, and how often piles are reconfigured (or turned) during the composting process. Aerated static piles are typically larger than windrows but are turned far less often.

1.3.4. Positively Aerated Systems

These systems use a blower to introduce air into the composting mass. In most cases positively aerated systems do not use a membrane cover but there are two or three vendors selling positively aerated membrane-cover systems. The most prominent of these is Gore, but AG-Bag (Poly Flex) and Composting Systems International (CTI) also offers positively aerated systems with a membrane cover. In the case of Poly Flex and CTI, the membrane is a polyethylene material. In the case of Gore the membrane is a proprietary “breathable” membrane.

1.3.5. Negatively Aerated Systems

The predominant advantage of negatively aerating a static pile is that the exhaust can be directed to a point source and put through a control system. Typically the control system is a biofilter. Negatively aerated static pile systems can be insulated by using a layer of woody organic material (like wood chips) or a membrane. Engineered Composting Systems offers a negatively aerated static pile system with an innovative inflatable form, which the pile is built over before it is removed.

1.3.6. Hybrid Systems

Some facilities utilize a combination of the systems described above. Some systems use an enclosed drum for the initial composting. This is typically followed by a more traditional aerated phase.

The type of system and how it is operated relies on a number of factors, the feedstock, the goal of the facility, the available land area, project budget, climate, and other factors.

Table 1 - Summary of Composting Systems

Composting System	Advantages	Disadvantages	% of market	Complies with EPA Pathogen Reduction Requirements	Cost	Time
Passive Piles	• Low cost • Inexpensive equipment • Easy to operate	• Little odor control • Little process control • Land intensive • Difficult to monitor • Not good for all feedstocks	<5%	No	Low	Many months to years
Turned windrows	• Relatively low cost • Easy to operate • Easy to monitor • Accepts a wide variety of feedstocks	• Land intensive • Limited process control • Limited odor control • Can be water intensive • High labor costs	90%	Yes	Low - Medium	2 months – 1 year, depending on management
Aerated Static Piles	• Moderate to good process control • Lower area requirement • Relatively low labor	• Need a homogenous initial mix • Requires engineering • Difficult to adjust once built • Need access to utilities	<5%	Yes	Low -High	2 - 4 months
Hybrid Systems	• Depends on system, can have best of both systems • High degree of process control	• Requires engineering • Sophisticated to operate • Need access to utilities	<5%	Yes	High	Varies

1.3.7. California Compost Facilities

The California Integrated Waste Management Board conducts periodic studies to gauge the status of the compost and mulch-producing infrastructure in California. The most recent study (CIWMB 2009) documented the presence of over 230 operating facilities processing organic materials in California. Although the survey is comprehensive, it is not exhaustive. The survey is focused on permitted composting facilities (both at the Notification and Full permit tier) and excludes facilities that are excluded from permitting requirements and those that do not handle materials commonly disposed of in landfills[1].

The 2009 Survey documented over 9 million tons of organics being processed by CA facilities. However, not all of this is composted. Figure 1 (CIWMB, 2009) shows the feedstock sources for all types of facilities (including stand-alone composting facilities but also chipping & grinding operations at landfills, material recovery facilities and transfer stations).

Figure 1 – Feedstock Sources for California Compost Facilities

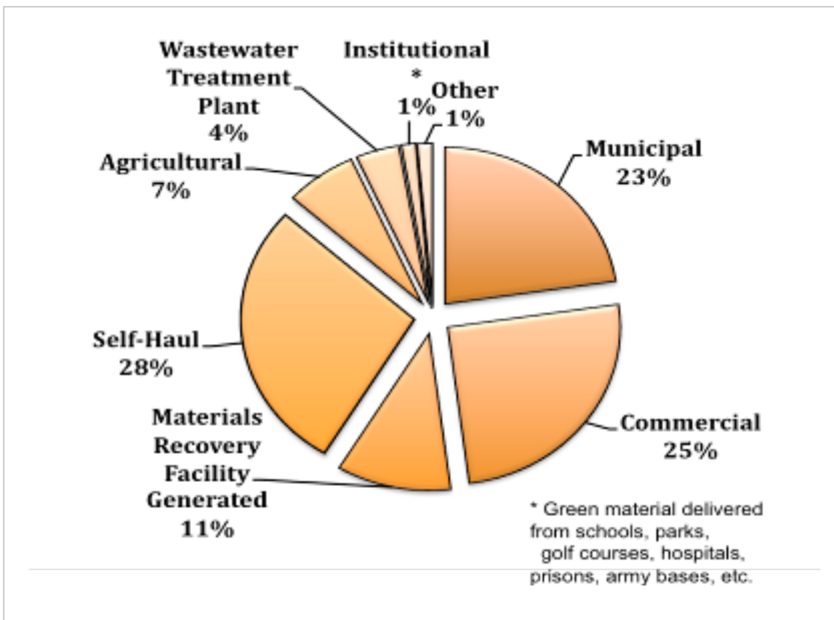
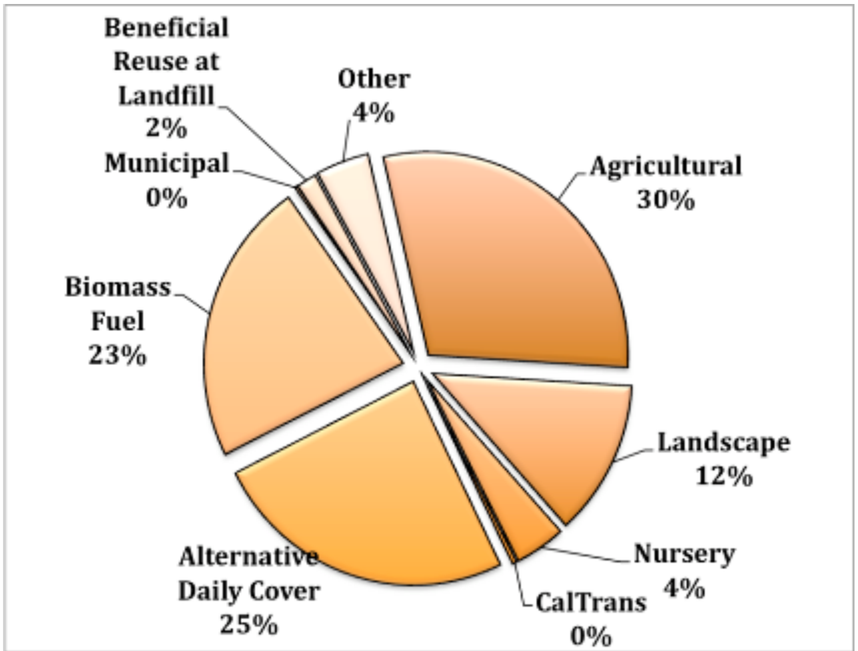


Figure 2 (CIWMB, 2009) shows the percentage of products sold by all facilities to markets in 2008.

Figure 2 - Percentage of Products Sold by All Facilities to Markets in 2008

[1] In some jurisdictions, Local Enforcement Agencies permit manure handling facilities and mushroom farms, these were largely excluded from the 2009 Survey.



1.4. Compost Feedstocks

A range of organic waste materials are suitable feedstocks for compost production. The type and quantities of feedstocks will vary by location. The sources of typical common compost feedstocks and their source and common disposal practices are listed in Table 2. Several biomass inventories have been conducted including inventories of agricultural wastes in California and biomass in New Jersey and Washington State (Brennan, 2007; Frear et al., 2005; Matteson and Jenkins, 2007). Inventories have also been conducted to characterize waste at landfills in California (Cascadia Consulting Group, 2004).

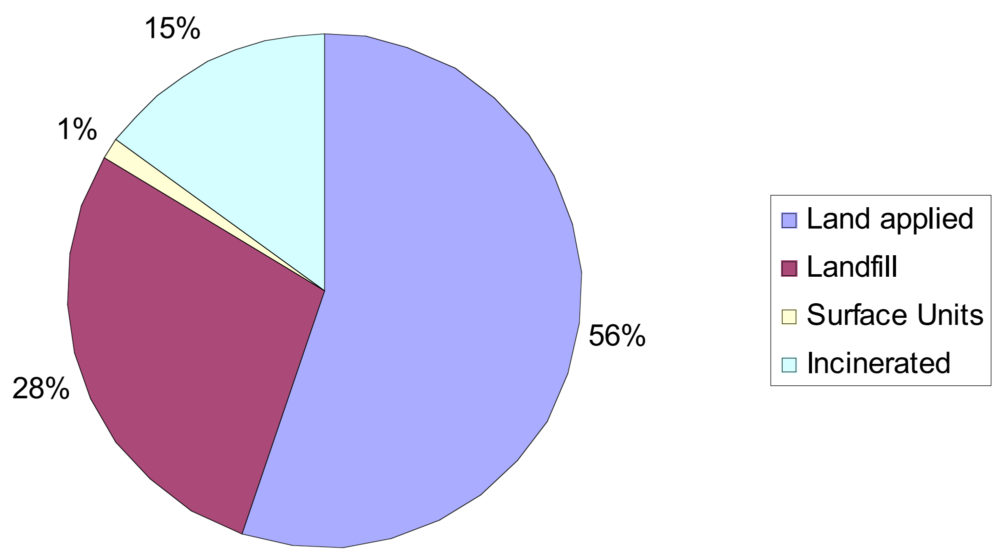
Table 2 – Feedstocks Typically Composted

Material	Source	Composition	Typical Disposal Practice	Composted?
Yard Trimmings	Municipal and commercial collection programs,	Leaves, grass, brush, etc.	Landfilled	Yes
Food Scraps (Residential + Commercial)	Municipal collection programs	Residential food scraps, restaurants, supermarket, etc	Landfilled, Sent to sanitary sewer	Yes
Food Scraps	Industrial	Food processing waste, production waste	Cattle feed, Directly applied to land	Yes
Manure	Feedlots, animal rearing facilities	Excrement, urine and bedding from a variety of animals	Directly applied to land, stored in lagoons	Yes
Biosolids	Municipal wastewater treatment plants	Solids from wastewater treatment	Directly applied to land	Yes
Liquid wastes	Industrial facilities	Waste liquids from	Directly applied	Yes

Material	Source	Composition	Typical Disposal Practice	Composted?
		industrial processes	to land, sent to sanitary sewer	
Mixed solid waste	Municipal and commercial collection programs	Mixed residential and commercials waste	Landfilled	Yes

Data collected from state regulatory agencies, US EPA and individual wastewater treatment facilities indicate that 7,180,000 dry U.S. tons of biosolids were produced in 2004. Of that, approximately 55% were land applied. The remaining 45% were disposed of in MSW landfills, surface disposal units and/or incineration facilities. Of this 45%, approximately 63% were disposed of in landfills, 4% were placed in biosolids only surface disposal units and 33% were incinerated. Figure 3 (Beecher et al, 2007) shows the disposal methods for biosolids in the U.S. in 2004.

Figure 3 –Biosolid Disposal Methods in the U.S.



Studies by the U.S EPA (2007) identify that 31.7 million tons of post-consumer food waste was generated in 2007 in the U.S., or 12.5% of total national MSW waste generated. An estimated 32.6 million tons of yard trimmings were generated in 2007, or 12.8% of total national MSW generated. Table 3 illustrates the percentages of post-consumer solid food waste and yard trimmings generated in the U.S. in 2007.

Table 3 - Tons of Post-Consumer Solid Food Waste and Yard Trimmings in the U.S.

2007 Figures (million tons/%)	Food Waste	Yard Trimmings	TOTAL
MT Generated in 2007	31.7	32.6	**64.3**
% of MSW Generated in 2007	12.5%	12.8%	**25.3%**

EPA further identified that 64.1% of yard trimmings generated were recovered, and 2.6% of food or other organic waste was recovered (including for composting). This is summarized in the table below.

Table 4 - EPA Estimates for Organic Waste Recovery in U.S. in 2007 (million tons)

Material	Weight Generated	Weight Recovered	Recovery As a Percent of Generation
Paper and paperboard	83.0	45.2	54.5%
Glass	13.6	3.22	23.7%
Metals			
Steel	15.6	5.28	33.8%
Aluminum	3.35	0.73	21.8%
Other nonferrous metals*	1.76	1.22	69.3%
Total metals	*20.8*	*7.23*	*34.8%*
Plastics	30.7	2.09	6.8%
Rubber and leather	7.48	1.10	14.7%
Textiles	11.9	1.90	15.9%
Wood	14.2	1.32	9.3%
Other materials	4.43	1.16	26.2%
Total Materials in Products	*186.1*	*63.3*	*34.0%*
Other wastes			
Food, other**	31.7	0.81	2.6%
Yard trimmings	32.6	20.9	64.1%
Miscellaneous inorganic wastes	3.75	Neg.	Neg.
Total Other Wastes	*68.0*	*21.7*	*31.9%*
TOTAL MUNICIPAL SOLID WASTE	254.1	85.0	33.4%

Includes waste from residential, commercial, and institutional sources.
* Includes lead from lead-acid batteries.
** Includes recovery of other MSW organics for composting.
Details may not add to totals due to rounding.
Neg. = Less than 5,000 tons or 0.05 percent.

Approximately 60% of homes in the U. S. have kitchen disposal units. These rates are based on U.S. Census surveys (Gaia Consulting, personal communication). Units are more common in areas with new construction as they generally came into use during the post WW II housing boom. A select list of cities with % households having kitchen disposal units is shown in Table 5 below.

Table 5 – Percentage of Kitchen Disposal Units in U.S. Cities

City	% with Kitchen Disposal Units
Anaheim/ Santa Ana	94%
Atlanta	53%
Dallas	79%
Houston	72%
Los Angeles	73%
Milwaukee	52%
New York/ Long Island	6%

City	% with Kitchen Disposal Units
Philadelphia	43%
San Francisco/ Oakland	55%

Municipalities will encourage or discourage use of units based on the capacity of the wastewater treatment system to treat the excess oxygen demand associated with disposal of food wastes into the municipal wastewater system. For example, units in New York City became legal in 1997. Washington D.C. discourages use of units as their wastewater system is currently near capacity and kitchen disposal would overload their system.

For homes with units, there are questions about the fraction of food waste that is actually put down the kitchen disposal instead of put for collection with traditional MSW. Gaia Consulting (personal communication) estimates that 50% of all food waste is put into kitchen disposal units for homes with units. It is also likely that kitchen units are only used for certain types of food waste. For example, excess food scraps from plates may be disposed of using the kitchen unit whereas food scraps from food preparation may be disposed of in the trash.

The Washington State Department of Ecology conducted a County by County survey of different organic residuals. Results from two counties are shown in Table 6 below. King County is a primarily urban County with the highest population density in the state. Yakima County is a primarily agricultural County with the highest population of dairy cattle in the state as well as a significant fruit tree industry.

Table 6 – Organic Waste Generation by Location

Organic Waste Type	King County (dry tons)	Yakima County (dry tons)
Corn stover		10,200
Mint slugs		36,700
Hops residue		4,300
Dairy manure	24,,400	115,200
Cattle manure	4,700	43,850
Horse manure	26,900	30,200
Poultry manure	300	22,700
Land clearing debris	70,000	2,400
Cull fruit		17,000
Apple pomice		10,100
Grape/fruit pomice		11,000
Cheese whey	2,400	11,300
Beef processing	600	7,000
Animal mortalities	150	1,200
Fish waste	650	
Food waste	67,000	7,100
Yard non wood	147,000	22,000
Other organics	15,500	800

Organic Waste Type	King County (dry tons)	Yakima County (dry tons)
Paper	729,000	7,8500
Grease	11,500	1,400
Biosolids	30,000	2,200
Total (dry tons)	**1,130,100**	**435,150**

The New Jersey survey separated waste materials into categories that were chosen based on a bioenergy focus (Brennan, 2007). The categories include:

- Sugars/starches
  - Agricultural crops suitable for fermentation including food processing residues
- Lignocellulosic biomass
  - Clean woody material from the MSW stream including yard debris, forestry waste and pallets
- Bio-oil
  - Waste oils and oil crops suitable for biodiesel
- Solid wastes
  - Biomass component of MSW, Construction and demolition, food wastes and soiled paper, all classified as primarily lignocellulosic biomass
- Other wastes
  - A general other category including animal wastes and biogas from wastewater treatment and landfills

The New Jersey inventory estimates that 75% of the biomass is derived from municipal solid waste and is concentrated in areas of high population density. The inventory does not provide any additional details on quantities of specific types of waste.

Matteson and Jenkins (2007) analyzed an inventory of food residues in California, including both residues generated by producers and processors as well as by consumers. Information for the survey was gathered from a range of sources including producers, processors and State publications (CIWMB, direct surveys, a study by von Bernath et al., 2004 and the California Biomass Facilities Reporting System available at http://biomass.ucdavis.edu/). Quantities of food residuals, along with the authors' estimates of the available fraction for each category are shown in Table 7 below. The available fraction of waste is defined by the author as quantity that might be technically available considering the constraints of access, handling and costs.

Table 7 – Food Residuals in California

Category	Gross resource dry Mg/y	Available fraction %	Available resource dry Mg/y
Vegetable crop	1,098,477	8	91,187

Category	Gross resource dry Mg/y	Available fraction %	Available resource dry Mg/y
Food processing			
High-moisture	207,703	65	135,007
Low-moisture	433,377	75	326,577
Meat processing	65,304	70	45,713
Grain and fiber processing	454,170	80	363,336
Food waste in MSW (landfilled)	1,676,650	50	838,325
Food waste in MSW (diverted)	295,879	50	147,940
Total	**4,231,561**	**46**	**1,948,085**

The Cascadia Consulting Group surveyed waste materials at landfills in California, dividing the State into 5 regions. Waste material was collected at landfills from both commercial and residential sources. The material was then sorted in to groups. Organics potentially suitable for composting comprise a majority of the materials in the 10 most prevalent types. The % and weight of organics in MSW according to this survey are shown in Table 8 below.

Table 8 – Survey of Waste Materials in Landfills in California

Material Type	Estimated percentage	Estimated tons
Food	14.6	5,854,352
Lumber	9.6	3,881,214
Uncoated Corrugated Cardboard	5.7	2,312,147
Remainder/composite paper	5.7	2,274,433
Remainder/composite organics	4.4	1,752,803
Leaves and grass	4.2	1,696,022
Other miscellaneous paper	3.5	1,400,526

It is important to note that animal waste and agricultural residues were not present in the organics that were inventoried. This is likely due to the fact that these materials are not routinely landfilled. However, agricultural residues and animal manures make up a significant portion of total potentially compostable organics and should be considered in an inventory of compost feedstocks.

2.0 EXISTING QUANTIFICATION METHODOLOGIES

This section provides a summary of existing quantification methodologies for reducing methane emissions from organic waste diversion from a landfill to a compost facility.

This section seeks to answer the following questions:

- Are methodologies appropriate for use in the United States?
- Are methodologies based on modeling, emission factors or direct measurement?

2.1. Clean Development Mechanism

2.1.1. About

The United Nations Framework Convention on Climate Change (UNFCCC) Clean Development Mechanism (CDM) is a market based methodology to help countries reduce their emissions and engage the private sector to contribute to emissions reduction efforts. This protocol addresses project activities where fresh waste (i.e. the organic matter present in new domestic, and commercial waste/municipal solid waste), originally intended for landfilling, is treated either through one or a combination of the following process: composting, gasification, anaerobic digestion, refuse derived fuel (RDF) processing/thermal treatment without incineration, and incineration (Clean Development Mechanism, undated, 2007 and 2008).

Quantification of the baseline emissions includes the following parameters:

- CH_4 – Net emissions from decomposition of waste at the landfill (methane generated minus methane destroyed in the absence of Project Activity). Methane production for a given year is calculated using a first order decay model.
- CO_2 – Emissions from electricity consumption from the grid or generated onsite/offsite
- CO_2 – Emissions from thermal energy generation if applicable

Assuming diversion of organics to an active aerobic composting facility, quantification of the Project Activity emissions includes the following parameters:

- CO_2 – On-site fossil fuel consumption due to project activity other than for electricity generation (e.g. vehicles used on-site, heat generation, etc.)
- CO_2 – Emissions from on-site electricity consumption
- N_2O, CH_4– Direct emissions from the waste treatment processes, applicable for composting activities. The composting process may not be complete and could result in anaerobic decay.
- CO_2 emissions from the decomposition of biomass are not accounted as GHG emissions.

During storage of organic waste and application of compost, N_2O emissions might be released. A study conducted of the composting process showed a total loss of 42 mg N_2O -N per kg composted dry matter can be expected (from which 26.9 mg N_2O during the composting process) (Schenk et. al. , 1997). Based on these values, the default emission factor of 0.043 kg N_2O per tonne of compost is used to calculate emissions.

Although the bulk aerobic compost systems maintain adequate oxygen availability, areas of anaerobic activity may occur. In addition, the end-use of the compost may be applied under anaerobic conditions. This methodology conservatively assumes a residence time for the compost in anaerobic conditions equal to the crediting period. Though located in a compost pile, this anaerobic digestion is treated identically to waste that had decayed anaerobically in a landfill. Through established air sampling procedures, waste that degrades under anaerobic conditions (oxygen content <10%) is counted toward the methane generation for a given compost operation.

2.1.2. Is methodology appropriate for use in U.S?

The Clean Development Mechanism protocol was created for use in developing countries. It currently has several certified projects, all of which are located in developing nations. The baseline developed for this protocol considers landfills as unmanaged open dump sites. This suggests that conditions in these sites in comparison to sanitary landfills that are characteristic of the U.S. would be dissimilar. Open dump sites would likely maintain aerobic conditions for a longer period. Once anaerobic decomposition has begun in these landfills, the absence of gas collection systems would likely result in higher CH_4 emissions than in managed sites. As open sites, the influence of local climate would likely be significant. In comparison, the use of clay liners, and the large volume and forced compaction of waste in sanitary landfills would result in more rapid generation of CH_4 that is less sensitive to external variability in climate. In addition, composting in developing countries is likely to be less intensively managed than composting facilities in the U. S.. For all facilities that treat biosolids in the U. S. , time and temperature requirements are in place. Similar requirements are in place in California for all composting facilities and in Washington State for all facilities that accept food scraps. In addition, many states require permits for compost facilities that include plans for runoff management. Odor is a major concern for composting facilities in the U.S., with odor problems developing primarily when compost piles become anaerobic. Process to control odors will also reduce the potential for fugitive emissions.

2.1.3. Is methodology based on modeling, emissions factors or direct measurement?

In both the baseline and the project activity scenario, this quantification methodology uses an algebraic model to account for variables in the potential and actual emissions. Among others, variables include factors to account for biogas capture at the landfill, degradation rates for different organic feedstocks and climate correction variables. Quantification of the Project Activity scenario incorporates a sampling methodology to determine how much methane is generated in the anaerobic sections of the compost pile.

2.2. Chicago Climate Exchange

2.2.1. About

Noting that significant quantities of methane are generated from decomposing organics in landfills, the Chicago Climate Exchange (CCX) published draft guidelines for methane avoidance projects. In general, qualifying projects are composting operations, which divert food, yard waste or biosolids from the landfill and into a composting operation. The quantification methodology compares the baseline scenario with the project scenario incorporating monitoring & verification data as described below (Chicago Climate Exchange, 2008):

The baseline scenario is the situation where organic matter is left to decay anaerobically within the project boundary and methane is emitted to the atmosphere. The baseline emissions are the amount of methane emitted from the decay of the degradable organic carbon in the biomass solid waste composted in the project activity. It is assumed that, methane generated from a batch of deposited waste is emitted unabated to the atmosphere for up to five years from the time of deposition until captured by a gas collection system. The CCX methodology conservatively assumes that landfill gas capture and combustion systems are installed at the three year mark. Baseline methane emissions are therefore defined as those emissions that would have occurred without gas collection in the first three years following solid waste disposal, and with gas collection systems in place for the following seven years thereafter. A default gas collection efficiency of 75% is assumed.

The annual methane generation potential for the solid waste is calculated using a first order decay model that is specific to the type of materials diverted and other parameters. The results of the model shown in Table 9 below show the annual and 10 year cumulative carbon equivalent per each ton of organic waste diverted from landfilling.

Table 9 - CCX Projected Yields of Waste Streams Diverted from Landfilling

	Year 1	Year 2	Year 3	Year 4	Year 5	Year 6	Year 7	Year 8	Year 9	Year 10	Total
Waste Type	**(CO_2e/wet ton waste diverted)**										
Food Waste	0.278	0.231	0.192	0.040	0.033	0.028	0.023	0.019	0.016	0.013	**0.872**
Yard Waste	0.108	0.098	0.088	0.020	0.018	0.016	0.015	0.013	0.012	0.011	**0.400**
Biosolids	0.048	0.040	0.033	0.007	0.006	0.005	0.004	0.003	0.003	0.002	**0.150**

For the project scenario, the CCX requires accounting of project activity and incremental fossil fuel emissions increases. If it is estimated that the project related fossil fuel use CO_2 emission are greater than 5% of those in the baseline, the fossil fuel use and CO_2 emissions of the project are not de minimis and must be included. The following lists the sources of CO_2 emissions due to incremental transport distances that must be considered:

- The collection points of biomass and the composting site as compared to the baseline solid waste disposal site,
- When applicable, the collection points of wastewater treatment site and composting site as compared to baseline solid waste disposal site,
- Composting site and the soil application sites.

Fossil fuel based energy and CO_2 emissions used by the project activity facilities includes energy used for aeration and/or turning of compost piles/heaps and chopping of biomass for size reduction and screening of the compost product.

2.2.2. Is methodology appropriate for use in U.S?

This methodology was developed for use in the U. S. Baseline values were developed using sanitary landfills as a point of reference. Baseline values for methane emission from landfills were developed considering current US regulations as well as US EPA estimates of system efficiencies. The committee used current U.S. EPA regulations on gas capture as the basis for determining gas collection efficiency during the initial phase of landfilling. Current regulations require gas collection systems to become operational between 2 to 5 years from the time where waste is deposited in a cell. The committee agreed on a 3 year time period of 0 gas collection efficiency based on the fact that gas collection was not legally required up until this point. The committee used the US EPA default values for gas collection efficiency (75%) from the point when collection systems become operational. The committee also opted for process controls during the composting process as a viable alternative to default factors for fugitive emissions. US EPA time and temperature requirements for pathogen destruction are well understood industry standards that are likely to significantly reduce process emissions. The committee adopted these requirements for all eligible projects in lieu of direct measures or default emissions factors.

2.2.3. Is methodology based on modeling, emissions factors or direct measurement?

For the baseline scenario, the CCX methodology uses a model to determine emissions factors for anaerobic waste decomposition as shown in Table 9. For project activity including transport emissions and electricity consumption, CCX relies on a series of emissions factors including those provided by the IPCC and the World Resources Institute. No direct measurement or sampling is required.

2.3. Alberta Aerobic Composting Protocol

2.3.1. About

In general, the Alberta Aerobic Composting Protocol (AACP) applies to avoided methane emissions from materials anaerobically decomposed in landfills and covers the diversion of organic residues from landfill for biological decomposition to a condition stable for nuisance-free storage and safe use in land application. The organics materials considered include agricultural and agri-food residues, the organic portion of MSW, food wastes, forestry and landscaping wastes, etc (Alberta Environment, 2008). The potentially relevant carbon sources and sinks for the baseline as well as the project scenarios are defined as follows in the offset quantification:

$$\text{Emissions}_{\text{baseline}} = \text{Emissions}_{\text{Decomposition and Methane Collection/Destruction}}$$

$$\text{Emissions}_{\text{Project}} = \text{Emissions}_{\text{Facility Operation}} + \text{Emissions}_{\text{Material Treatment}} + \text{Emissions}_{\text{Decomposition and Methane Collection/Destruction}} + \text{Emissions}_{\text{Fuel Extraction and Processing}}$$

Each of these equations has one or more equations that define and sum the relevant emissions from each stage. Calculation of these emissions levels rely on a series of standard emissions factors for fuel extraction and production as well as combustion/use for CO_2, methane, and N_2O. Since the emission factors are linear, there is no time element incorporated into the result. However, emissions reduction offsets can be generated for a maximum period of 8 years.

This protocol also provides some flexibility based on geographic or site specific circumstances. For example, the methane generation potential (kg CO_2/ton waste) variable must be selected based on the Canadian province. Furthermore, if site landfill or wastestream emissions are documented, those can be substituted into the emissions equations as well.

2.3.2. Is methodology appropriate for use in U.S?

The Alberta protocol is modeled closely on the CDM protocol. It has been modified to reflect Province specific waste inventories as well as types of landfills commonly found in Canada. A wide range of organic wastes are grouped together in this protocol. The protocol is not acceptable for use in the U. S. as it does not provide specific decay rate constants for different feedstocks. In many cases in the U. S., yard waste is banned from landfills. Quantities and characteristics of organics diverted to composting facilities will not reflect the impact of these bans on waste composition in the U. S. In addition, the Alberta protocol's grouping of landfills is not compatible with landfills in the U. S. The guidelines do not reflect gas collection requirements in the U.S., stages of methane emissions, or the predominance of large-scale sanitary landfills.

2.3.3. Is methodology based on modeling, emissions factors or direct measurement?

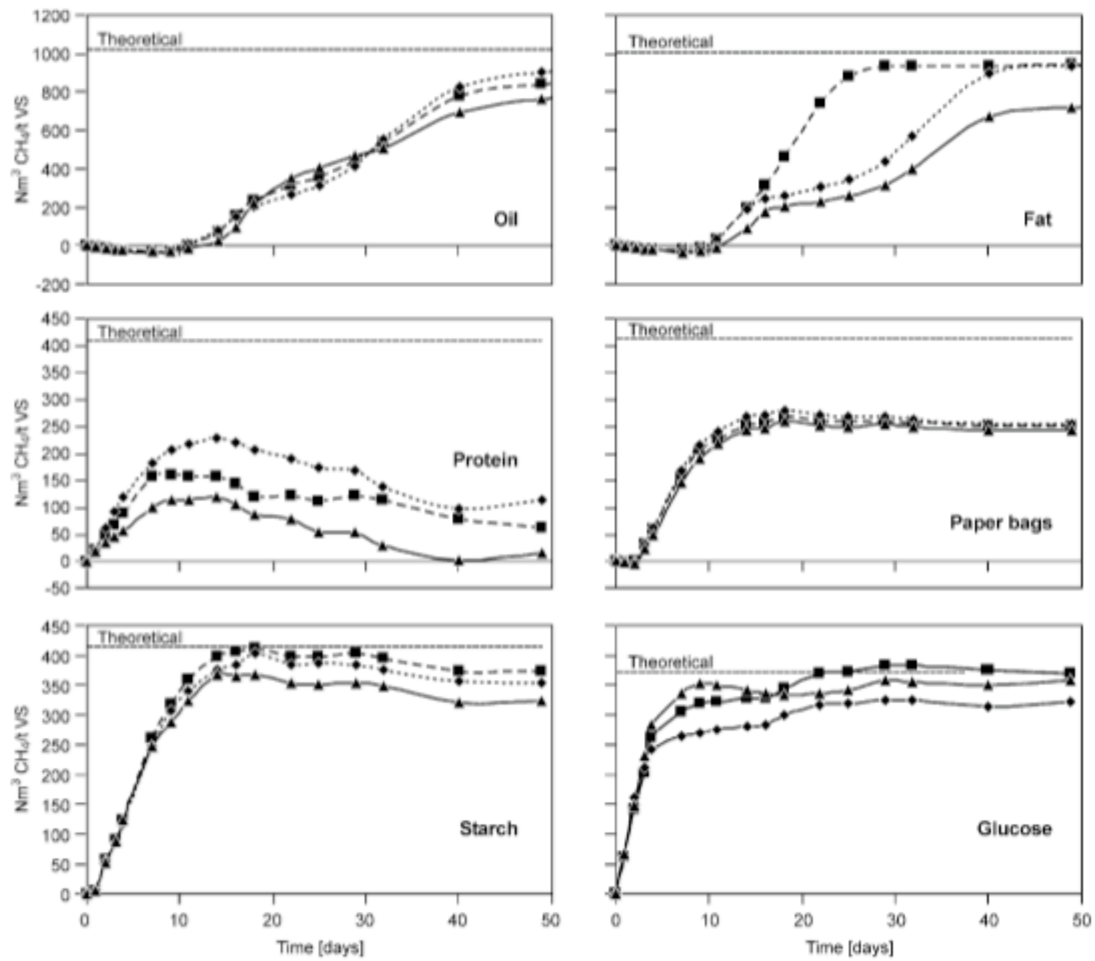
The Alberta emissions model is highly dependent on a few variables including Degradable Organic Carbon (DOC), which varies based on Province and the type of landfill. Direct landfill specific emission measurements can be substituted if they exist. Project Activity emissions are based on two compost emission factors for CO_2 and N_2O. In addition, other factors can be applied such as the carbon sequestration potential within the compost, and methane/N_2O emissions from the use of compost.

2.4. Other

2.4.1. Volatile Solids

Volatile solids is straight forward and inexpensive to measure and may be used as an indicator of the CH_4 generation potential for different substrates (US EPA, 2001) used in composting facilities. Hansen et al. (2004) developed a lab scale incubation to evaluate the CH_4 generation potential of different waste materials. The procedure requires a 50 day incubation and was tested on mixed materials as well as on individual waste materials. Total volatile solids destruction was considered to be the theoretical limit of CH_4 production. The actual and theoretical CH_4 generation potential for several feedstocks is shown in Figure 4 below (Hansen et al., 2004).

Figure 4 – CH_4 Generation Potential using Volatile Solids



In this study, source separated organic household waste had a methane potential of 495 ml CH_4/g VS. For individual components of MSW, paper bags, starch and glucose yielded 63, 84 and 94% of the predicted CH_4 potential.

For materials with chemical composition similar to food scraps, a maximum VS destruction rate of 80% can be used as a baseline to calculate CH_4 generation potential. For other materials, it would be necessary to conduct bench scale decomposition studies to determine the CH_4 generation potential.

3.0 SCIENTIFIC UNCERTAINTY

This section discusses the state of scientific understanding around methods for quantifying emissions reductions associated with diversion of organic waste from landfills to composting. It seeks to answer the following questions:

- How accurate are existing measurement or quantification methods?
- What measurement options are available and what are their associated costs?
- What are the likely ranges of uncertainty?

3.1. CDM and CCX

3.1.1. How accurate are existing measurement or quantification methods?

The equation used to calculate GHG reductions for the CCX protocol was based on the equation used in the CDM protocol is illustrated in Figure 5.

Figure 5 - CDM CCX GHG Reduction Equation

$$BE_{CH4,SWDS,y} = \varphi \cdot (1-f) \cdot GWP_{CH4} \cdot (1-OX) \cdot \frac{16}{12} \cdot F \cdot DOC_f \cdot MCF \cdot \sum_{x=1}^{y} \sum_j W_{j,x} \cdot DOC_j \cdot e^{-k_j \cdot (y-x)} \cdot (1-e^{-k_j})$$

Where:

$BE_{CH4,SWDS,y}$	= Methane emissions avoided during the year y from preventing waste disposal at the solid waste disposal site (SWDS) during the period from the start of the project activity to the end of the year y (tCO_2e)
Φ	= Model correction factor to account for model uncertainties (0.9)
f	= Fraction of methane captured a the SWDS and flared, combusted or used in another manner (zero for the first three years)
GWP_{CH4}	= Global Warming Potential (GWP) of methane, valid for the relevant commitment period (21)
OX	= Oxidation factor (reflecting the amount of methane from SWDS that is oxidized in the soil or other material covering the waste (0.10)
F	= Fraction of methane in the SWDS gas (volume fraction) (0.5)
DOC_f	= Fraction of degradable organic carbon (DOC) that can decompose (from Table 1)
MCF	= Methane correction factor (1)
$W_{j,x}$	= Amount of organic waste type *j* prevented from disposal in the SWDS in the year *x* (metric tons) (monitored)
DOC_j	= Fraction of degradable organic carbon by weight in the waste type *j* (from Table 1)
k_j	= Decay rate for the waste type *j* (from Table 1)
j	= Waste type category (index)
x	= Year during the crediting period: x runs from the first year of the first crediting period (x = 1) to the year y for which avoided emissions are calculated (x = y)
y	= Year for which methane emissions are calculated

The CDM equation was developed for use in developing nations with the assumption that landfills in these nations are not well managed. Landfills were assumed to be smaller in scale and closer to open dump sites than to sanitary landfills. The CCX protocol was developed for use primarily in the U. S. where sanitary landfills predominate. During the protocol development process the committee focused on a number of factors in this equation and attempted to adapt values from the CDM specified defaults to be more reflective of the type of landfills that are characteristic of the United States. The primary uncertainties related to the development of the CCX protocol centered on understanding conditions within sanitary landfills and decomposition rates of individual feedstocks under those conditions.

The most critical factor controlling the methane avoidance credit given for diversion of a particular feedstock is the decay rate (identified as k rate units in the above equation) constant for that feedstock. The CDM protocol gives decay rate constants for different feedstocks based on the climate at the landfill. Climates are divided into 4 categories:

- Warm dry
- Warm wet
- Cold dry
- Cold wet

Both feedstock specific characteristics as well as conditions within landfills will determine how much and how quickly methane will evolve from anaerobic decomposition of different substrates. The primary factors related to the landfill environment that affect decomposition rate include temperature and moisture concentrations within the individual landfill cells. The committee focused on many areas of uncertainty re landfill environment. These included temperature and moisture within the landfill. Members of the committee argued that sanitary landfills have an internal climate that is dependant on waste characteristics and independent on the external environment. A decision was made to use a single decomposition rate for each feedstock.

Decomposition rates of particular waste streams will vary based on landfill environment as well as feedstock characteristics. Feedstock characteristics include the % solids of the feedstock, nutrient content, particle size, and chemical compounds degradability under anaerobic conditions. For first order decay rate constants for specific feedstocks the committee used the CDM guidelines as a basis. Data from peer review literature was also used. The committee agreed that there was an over reliance on a single study: Eleazer et al. (1997) to determine the decay rates. In addition, the conditions of the study were not reflective of a landfill environment. The committee also used methane generation potential and decay rates from anaerobic digestion studies as away to develop alternative k rates. The committee also discussed the degree of heterogeneity of materials within particular categories. Food waste was thought to be more homogeneous than yard waste. This was an area of high uncertainty in the protocol development process with regards to environment within the landfill and decay rates of individual waste components.

Secondary factors that influenced the credit for landfill diversion include the % of CH_4 that is oxidized by landfill cover soils and efficiency of gas collection systems. The first factor that the group focused on was gas collection efficiency. The US EPA has specified a 75% efficiency in gas collection over the life of a landfill. During the course of the protocol development, the EPA hosted a workshop to reevaluate several assumptions that were used in the WARM model including the 75% collection efficiency. At the workshop it was decided that the efficiency of gas collection more than certainly varied over the different stages of the landfill with higher efficiency post closure while collection systems were operating and minimal efficiency during the early phases of landfilling or at the open face of the landfill before collection systems were in place. The IPCC gives a general efficiency for gas collection at landfills from 40-50%. The committee opted to limit the protocol to highly putrescible wastes (paper was not included). There was general agreement that collection efficiency could be set at 0% before collection systems were in place. There was substantial disagreement and uncertainty about gas collection system efficiency once the system was operational. The group decided to use the EPA default value of 75%. By focusing only on readily decomposable wastes and on the initial phase of landfilling agreement was reached on a 75% collection efficiency after a 3 year period and a 10% waste oxidation rate for cover soils.

Transportation emissions and process emissions were also discussed. It was generally agreed that these emissions would be minimal in comparison to the emissions reduced as a result of landfill diversion. It was agreed that these could be considered but in general, would be de minimis.

The group also discussed fugitive gas release during composting. Both process controls and potential default emissions factors were considered. It was noted that the CDM protocol has default factors for N_2O emissions. These were based on a limited review of the literature. The group used the survey of literature from the Brown et al. (2008) study as a guideline for both management practices to reduce emissions as well as to understand the range of fugitive gas emissions during the composting process. It was agreed that a process control that was inexpensive and straight -forward to implement would be the most effective. The committee recommended use of the US EPA CFR Part 503 requirements for pathogen reduction in biosolids compost for a performance standard. These are used for all compost facilities in California.

The Clean Development Mechanism protocol for methane avoidance for landfill diversion to composting operations takes a default debit for fugitive N_2O emissions during the composting process (Clean Development Mechanism, 2008). However, the anoxic conditions required for N_2O formation are potentially likely to occur in a landfill environment.

3.2. Alberta Protocol

3.2.1. How accurate are existing measurement or quantification methods?

The Alberta protocol for landfill methane avoidance for composting of organic residuals is loosely based on the CDM protocol as well as data collected by the Canadian government. In some ways it is more detailed than the CCX protocol but in others, it is greatly simplified in comparison to the CCX protocol. The protocol does not distinguish between different types of organic waste materials and is written to address 'agricultural and agri-food residues, the organic portion of MSW, food wastes, forestry and landscaping wastes, etc'. Decay rate constants are not provided for different types of waste materials. Methane generation potential is taken from IPCC default values. These derive from the degradable organic fraction of the grouped waste materials. Values are given by province rather than by waste type. Based on the information provided by the protocol there is no way to gauge how quickly material will decompose and so at what point the methane generation potential of the material will be exhausted. The protocol does not address potential differences in decomposition or gas collection efficiency over the course of the life of the landfill.

Baseline emissions are defined as the sum of the emissions under baseline conditions. Figure 6 illustrates the equation used to calculate CH_4 generation potential of the waste material.

Figure 6 - Alberta Protocol Methane Generation Potential Equation

$$L_0 = MCF * DOC * DOC_F * F * 16/12 * 1000 \text{ kg } CH_4/\text{t } CH_4$$

Where L_O = CH_4 generation potential (kg CH_4/ t waste)
MCF = CH_4 correction factor (fraction)
DOC = degradable organic carbon (t C/t waste)
DOC_F = fraction DOC dissimilated
F = fraction CH_4 in landfill gas
16/12 = stoichiometric factor

The values of each variable in the equation were taken directly from IPCC with the exception of DOC. DOC was taken from average values by province as determined by Environment Canada. The Lo varies from 90.3 (Ontario) to 117.0 (Yukon and the Northwest Territories). Differences in the Lo derive from differences in the DOC. This would suggest that conditions within the landfill are not considered to vary based on external climate.

There is very little discussion of environment within the landfill. The protocol indicates that the specific quantities of credits for a particular project will vary based on the landfill that the waste is diverted from. The protocol follows the CDM method of dividing landfills according to the depth of waste. Landfills are described as controlled or uncontrolled facilities. If the landfill has a methane collection system in place and operating the protocol specifies that this must be accounted for in determining the quantity of credits that the composting project is eligible for.

Variation in landfills is described according Table 10 below.

Table 10 – Alberta Protocol Variation in Landfills

Parameter	Mixed-Waste Landfills				Wood Waste Landfills
	Managed	Unmanaged – Deep (>= 5m waste)	Unmanaged – Shallow (< 5m waste)	Uncategorized	
Methane Correction Factor (MCF)	1.0	0.8	0.4	0.6	0.8[a]
Fraction of CH_4 in landfill gas (F)	0.5				
Fraction of degradable organic carbon dissimilated (DOC_F)	0.77				0.5
Fraction of degradable organic carbon (DOC)	See Appendix A				0.3

a - the default condition for a wood waste landfill is an unmanaged, deep landfill (Environment Canada, 2006). This parameter may be changed if the emissions are being calculated for an alternate type of wood waste landfill.

Total credits from diversion are based on the methane generated minus the methane collected and destroyed, depending on type of landfill. The location of the landfill had no bearing on the diversion credits provided. Type of landfill is the sole factor considered here.

The protocol also provides for default emissions factors for a range of variables. Transportation and energy use is included here. Fugitive emissions during composting are also included. The debits for fugitive emissions during composting are 0.004 kg CH_4 per kg biomass and 0.0003 kg N_2O per kg biomass. These emissions factors were taken directly from IPCC and do not include any provisions for on site emissions controls or management practices to reduce or eliminate emissions. Compost is considered cured if it meets the CCME standards for maturity and destruction of pathogenic organisms.

In general the Alberta protocol is a very simplified accounting document. It is highly conservative in the credits provided for composting facilities. It is based primarily on IPCC default factors and does not consider in detail how sanitary landfills will differ from unmanaged dumping sites. It does not distinguish between different types of waste materials, different types of composting operations, or different rates of decay in different types or climates for landfills.

4.0 ADDITIONALITY - PERFORMANCE STANDARD TEST

Project developers pass the Performance Standard Test by meeting a program-wide performance threshold – i.e. a standard of performance applicable to all composting projects, established on an *ex-ante*[2] basis. The performance threshold represents better than "business-as-usual" organic waste management. If the project meets the threshold, then it exceeds what would happen under the business-as-usual scenario and thus generates surplus/additional GHG reductions.

This section aims to answer the following questions:

- Does the project type easily lend itself to a standardized, performance based approach for estimating baselines and/or determining additionality?
- What are the current drivers (if any) behind technologies or practices that may reduce emissions?

4.1. OWD Protocol Performance Standards

For the Organic Waste Digestion protocol, the Reserve uses a performance based threshold that serves as a national "best-practice standard" for managing organic wastes.

Because of the potential for OWD projects to digest numerous feedstocks, the performance threshold defines those feedstocks that the Reserve has determined will generate surplus/additional GHG reductions if digested in a biogas control system – i.e. feedstocks that otherwise would have generated methane to the atmosphere under business-as-usual management conditions.

An organic waste digestion project therefore passes the Performance Standard Test if at least one of the following eligible organic waste streams is consistently, periodically, or seasonally digested in the project's biogas control system:

- Municipal Solid Waste (MSW) Food Waste: Food waste commonly disposed into a MSW system, consisting of uneaten food, food scraps, spoiled food and food preparation wastes from homes, restaurants, kitchens, grocery stores, campuses, cafeterias, and similar institutions.
- Agro-industrial Wastewater: Organic loaded wastewater from industrial or agricultural processing operations that, pre-project, was treated in an uncontrolled anaerobic lagoon, pond, or tank at a privately owned treatment facility. Excluded from eligibility based on the Reserve's performance standard analysis are wastewaters produced at breweries, ethanol plants, pharmaceutical production facilities, and pulp and paper plants.

Projects that co-digest organic waste together with manure must meet the OWD performance threshold as defined above to be eligible as an OWD project. Additionally, all livestock operations contributing manure to an OWD project must meet the eligibility requirements as

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[2] 'before the event'

defined in the most recent version (as of the time of project listing) of the Reserve's Livestock Project Reporting Protocol.

OWD projects may choose to digest multiple feedstocks, some of which may be ineligible per the Performance Standard Test. Ineligible waste streams may be co-digested alongside eligible organic waste streams, but will not result with surplus/additional GHG reductions. The Reserve defined this Performance Standard by analyzing organic waste management practices in the U.S. for numerous potential digester feedstocks. The eligible feedstocks as listed above represent the potential feedstocks that otherwise would have released methane to the atmosphere under business-as-usual management conditions.

4.2. Composting Performance Standards

4.2.1. Does the project type easily lend itself to a standardized, performance based approach for estimating baselines and/or determining additionality?

With the development of other protocols for composting facilities and the nature of the project activity, it appears that this type o f project does lend itself to a standardized, performance based approach for GHG reduction. This will be based on specific requirements such as:

- the feedstocks being composted at the facility and their traditional or baseline disposal method
- the operation of the composting facility and potential for the generation of GHG emissions

The primary rationale for the development of standardized performance thresholds is the 2007 EPA study summarized on Table 4. This study is quite recent, and well-researched, and can be used as a basis for a standardized approach for materials diverted away from landfills for reuse or composting.

The next step is to decide which waste streams are appropriate for inclusion in the standard, which ones aren't, and which ones need further study in specific areas. First, it is easy to recognize that MSW food scraps should be included as only 2.6% of the volume to landfill is recovered for reuse. These MSW food scraps are highly degradable in landfills and their diversion to compost facilities would be quite beneficial from a GHG emissions standpoint.

The next category of interest is wood waste. It is below 10% overall recovery/reuse, so it can be considered for inclusion, since reuse is not common practice and is significantly better than the BAU practice of landfilling. The CIWMB study cited in Section 1.3.7 shows that wood waste is in demand from both composters and processors, so there is a viable market for the material. However, the decay of wood in landfills appears to be very slow. The Ximenes et. al. 2008 cited below confirms the slow decay rate and low fraction of degradable organic carbon in wood. As noted in the study, two samples from landfills aged 19 and 29 years showed no significant loss of dry mass. Another sample of wood from a landfill aged 46 years lost 18% of its original carbon content. With no significant loss of carbon in 20 and 30 year old landfill-wood samples and a minor loss in the 46 year old sample, we can conclude that the decay rate for wood is much longer than project crediting periods and, according to this data, would not appear appropriate for inclusion unless further study shows that certain wood types decay within near term project crediting periods.

In the category of plastics, although the reuse and recovery rate of the material is low (6.8%), there doesn't appear to be a case for anaerobic decomposition of this material in landfills in project crediting timeframes.

The 64% recovery and reuse rate of yard waste materials appears to exclude this category from consideration in a performance threshold, since this is BAU practice. Some of this "reuse" is actually used as cover in landfills and would have a mix of aerobic and anaerobic degradation, however the extent of the anaerobic degradation for landfill cover is not clear at this time. For yard waste to be included in this performance standard, it would have to be shown show that a very high percentage (at least 80%) of this recovered yard waste is reused as daily cover and that this material decays in a predominately anaerobic manner. This study found no evidence to bear this out, so yard waste is not a candidate for inclusion in this performance standard at this time. In addition, a number of States have bans on this material going to landfills which is discussed in more detail in the regulatory additionality section. These bans reinforce the premise that the recovery and reuse of yard waste in the US reflects a BAU practice.

The 2007 EPA study did not address biosolids, even though some are sent to landfills. Typically biosolids are sent to landfills because they cannot be applied to agricultural lands or landfarmed for Federal or local regulatory reasons, so are not typically candidate materials for diversion. Figure 3 showed that just 28% of biosolids are sent to landfill in the US, with landfarming or land application being the predominant activity (56%). Thus, the BAU practice for biosolids from a national perspective is an essentially aerobic process of land application or landfarm. However, there may be some regions where land application or landfarming is not viable due to the lack of application areas to perform this activity. Further study would be needed to determine whether sending biosolids to landfills is the predominant activity for disposition of biosolids in certain regions. This would determine whether some areas could be included in a performance threshold for composting of this material.

4.2.2. What are the current drivers (if any) behind technologies or practices that may reduce emissions?

The following section outlines current drivers behind reducing emissions at composting facilities.

Environmental Requirements

In California and other states such as Washington, increased concerns about water quality are driving a trend towards aerated static pile composting. In addition, air quality emissions of VOC's and ammonia are also driving facilities to ensure their composting is aerated or aerobic, particularly in California.

Population Density

In less populated areas there is less pressure to conserve landfill space and to separately collect recyclables and/or compostables.

Costs

Composting involves receiving a material, cleaning and processing it into compost and selling the compost. This involves more processing steps (and thus more operating costs) then disposing of it in a landfill.

Another barrier is that to attract the waste stream, composters often charge less than the standard disposal rate (with hauling being equal). This need to under-price in order to get materials when in fact taking these materials results in higher operating and permit costs for a facility, is a major disincentive for compost operations to accept food waste.

In many states, particularly those without bans on yard trimmings from landfill, there are several barriers to implementing successful composting operations. Perhaps the most significant of these is the cost of landfilling, which is still relatively cheap in many states. Many municipal composting operations benefit from the avoided cost of disposal – the tipping fee for yard trimmings for example has to be lower than the tipping fee for mixed garbage, in fact it typically has to be low enough to encourage separation of the yard trimmings from the regular garbage and to justify a separate trip to a drop-off facility. In some cases if yard trimmings are collected at the local landfill or at a site located between the point of generation and the disposal site this cost differential doesn't have to be as great. Other barriers include:

Siting and Permitting

Composting facilities (like many solid waste handling sites can be difficult to site. Odors, land use compatibility, and traffic impacts are the most difficult of the potential issues to overcome.

For example, the major barrier to composting yard waste is related to the scale of the operation. For small- scale yard waste operations, it is relatively easy to stockpile leaves in a portion of a collection site. These will slowly decompose over a winter and be ready for use the following spring. The only equipment required for this type of operation would be a loader to pile the leaves. Most states do not regulate yard waste composting facilities. They typically go through a registration process, and where required, water quality monitoring. Depending on the state, there also may be capacity limitation (above a certain size, sites may need some sort of permit). If operators want to collect and process large quantities of yard waste, the major obstacle is citing a facility.

Food waste composting is most commonly done as an add- on to existing yard waste composting facilities. However, many institutions (universities, correctional facilities, resorts), will start up a food waste composting only project, typically using some small-scale composting technology. However the bulk of food waste composting is currently done at larger scale operations with food waste being one of the feedstocks that is composted. The primary feedstock at these facilities is likely to be yard waste although a small number of sites will also accept municipal biosolids or other materials.

The first major barrier for food waste composting is getting permitted to take food waste at a yard waste site. Some states make it easier than others, e.g., by starting out with a quantity limitation, or by only allowing pre-consumer vegetative food waste (i.e., not meats, dairy). Very quickly, composters who want to take all types and quantity of food waste will require a solid waste facility permit. Pennsylvania has created a General Permit that makes this less onerous, but that process is still challenging. In Illinois, food waste was recently redefined so that it is no longer considered as solid waste. This was done to make it easier to establish composting facilities that accept food waste. In general, regulations and permitting are the largest hurdles to establishing a food waste composting site.

Accepting food waste will also increased operating costs for running the facility. As food waste requires near immediate processing, staff and equipment has to be on site to accept and process materials when they are delivered. Appropriate process control and materials handling are critical to avoiding nuisance, odor and vector problems.

Separate Collection

In some cases separate collection (especially for food scraps) may be difficult to justify. In a few communities, particularly on the West Coast, food scraps are being added to the existing yard trimmings containers. This has proven to be a cost-effective tool in minimizing collection costs. This doesn't work out as well in states where yard trimmings generation is seasonal (the East Coast and Mid West), but food scraps generation is constant.

5.0 ADDITIONALITY – REGULATORY TEST

All GHG reduction projects are subject to a Regulatory Test to ensure that the emission reductions achieved by the project would not otherwise have occurred due to federal, state or local regulations. This section seeks to answer the question:

- How easy or hard might it be to distinguish between additional and non-additional projects?

5.1. OWD Protocol Regulatory Test

Under the Organic Waste Digestion Protocol, to maintain eligibility, a project must consistently digest at least one eligible waste stream. Ineligible waste streams may therefore be co-digested with eligible organic waste streams, but will not result with surplus/additional GHG reductions.

Project developers pass the Regulatory Test by demonstrating that:

- The project is digesting at least one eligible organic waste stream that is not required by any federal, state or local regulation, ordinance, or agency ruling to be diverted from a landfill (if solid waste) or an uncontrolled anaerobic wastewater treatment system (if wastewater)

For the OWD Protocol, it was determined that there are various state and local regulations, ordinances, and mandatory diversion targets that may obligate waste source producers or waste management entities to divert organic wastes away from landfills. An organic solid waste stream that is banned from landfilling, or has strong regulatory incentive to be managed in a system other than a landfill fails the Regulatory Test.

Although not required by federal regulation, California has mandatory landfill diversion targets that require a percentage of waste generated to be diverted from landfills to alternative management systems. Other states have non-mandated goals of a similar nature. This target may provide strong regulatory driven incentives to divert organic wastes from landfills. Thus, organic waste originating from a jurisdiction that has not yet met its landfill diversion target does not pass the Regulatory Test until the target is achieved.

Mandatory state diversion targets are not to be confused with state diversion goals. Should a state adopt a statewide waste diversion goal that does not impose penalties on jurisdictions for failing to meet diversion targets, then this state goal shall not result with a failure of the Regulatory Test.

Local jurisdictions may have bans on certain types of waste going to landfill, or may have mandatory ordinances that require the diversion of organic solid wastes from landfills. Should a local jurisdiction have a mandatory ban on food waste disposal at landfills, or otherwise have food waste diversion mandates, food waste streams originating from the jurisdiction would fail the Regulatory Test.

In some specific instances, however, there may be a valid exception to this rule if there is a local ordinance that was enacted for the purpose of generating carbon credits through waste diversion projects. Local municipalities could potentially enact mandatory food waste diversion ordinances or regulations with the intent of providing the necessary feedstock for a local OWD project. In these specific instances, the OWD project may be unable to source the necessary digester feedstock without the food waste diversion mandates. For the purposes of this protocol, a food waste stream produced at a jurisdiction requiring mandatory food diversion passes the Regulatory Test if:

- The OWD project digesting the food waste stream has an operational start date prior to or no more than12 months following the passage into law of the local food waste diversion mandate.

5.2. Relevant Regulations

For this task, the project team conducted an evaluation of existing and pending state and national regulations related to composting activities to determine if they are or may be required by regulation.

Table 11 shows a summary of state recycling goals, landfill bans on yard trimmings, and the number of permitted composting facilities. Note a goal implies a voluntary commitment, whereas a mandate requires a regulation in place.

Table 11 – Summary of Waste Regulations by State

State	State Recycling Goal/Mandate[3,4]	Yard Waste Ban[5] (Yes/No)	Estimated Number of Permitted Composting Facilities[6]
Alabama	25% goal in 1989 (the passing of HB 395 in '08 may up this rate)	No	Unknown
Alaska	25%	No	4
Arizona	No statewide recycling goal	No	10
Arkansas	50% goal by 2010	Yes	32
California	50% landfill diversion mandate	No.	150
Colorado	Governor's challenge of 50% by 2000	No.	29
Connecticut	40% goal by 2000	Yes	94
Delaware	70% goal by 2010	Yes (recent)	2 [4]
District of Columbia	45%		
Florida	75% goal by 2020	Yes	8
Georgia	25% goal by 1996	Yes	38
Hawaii	50% goal by 2000	No	5
Idaho	Non binding resolution	No	5

3 "Appendix 1, State Recycling Goals and Mandates" in "Processing and Marketing Recyclables in New York City May 2004 (Original source www.AFand PA.org), New York City Bureau of Waste Prevention, Reuse, & Recycling.

4 Personal Communication, Justin Gast, Resource Recycling, 2009.

5 BioCycle Magazine, State of Garbage in America 2008.

6 Compiled from various site web sources and published reports including BioCycle (2008). The definition of what constitutes a "facility" varies state to state. Most states do not permit agricultural composting facilities.

State	State Recycling Goal/Mandate[3,4]	Yard Waste Ban[5] (Yes/No)	Estimated Number of Permitted Composting Facilities[6]
	for 25% goal		
Illinois	25% by 1996 (The Illinois Solid Waste Planning and Recycling Act (415 ILCS 15)	Yes	40
Indiana	50% by 2001	Yes	122[7]
Iowa	50% by 2000	Yes	93
Kansas	No statewide goal	No	123
Kentucky	25% by 1997	*YW banned from some landfills	34
Louisiana	30%	No	?
Maine	50% by 1998	No.	80
Maryland	40% goal (1999)	Yes	5 [4]
Massachusetts	46% by 2000	Yes	223
Michigan	1998 policy encourages SWM percentages, 30%	Yes	155
Minnesota	50% (TC metro) 35% state (1989)	Yes	80
Mississippi	25% by 1996 (SN2984, 1991)	No	11
Missouri	40% by 1998 (SB530, 1990)	Yes	93
Montana	25% by 1996 (1991)	No	22
Nebraska	50% by 2002 (1992)	Yes/(Changed)	9
Nevada	25% goal by 1995 (AB 320, 1991)	No	4
New Hampshire	40% by 2000 goal	Yes	12
New Jersey	60% by 1996 (1992) 65% by 2000	Yes	172
New Mexico	50% by 2000 goal (SB 2, 1990)	No	29
New York	50% by 1997 (1987 SWMP)	No	35
North Carolina	40% by 2001 (1991)	Yes	120
North Dakota	40% by 2000 (1991)	No	8
Ohio	50% by 2000	*Some disposal restrictions on YW	444[8]
Oklahoma	Oklahoma State Recycling & Procurement Act	No	3
Oregon	50% by 2009 (1991)	No.	44
Pennsylvania	35% goal by 2005	Yes	465

[7] Indiana, Department of Environmental Management, Registered Yard Waste Composting Facilities, www.in.gov/idem/5077.htm#composting

[8] Ohio EPA, Licensed Class I, Class II, Class III and Class IV Composting Facilities, www.epa.ohio.gov/dsiwm/pages/comp_docs.aspx

State	State Recycling Goal/Mandate[3,4]	Yard Waste Ban[5] (Yes/No)	Estimated Number of Permitted Composting Facilities[6]
Rhode Island	70% (no deadline, 1989) SB 2797 (2008) sets municipal goals.	No	13
South Carolina	35% by 1995(Bill 3927, 1999)	Yes	96
South Dakota	50% goal by 2001 (HB 1001	Yes	128
Tennessee	25% by 2003 (HB 1252, 1991)	No	2
Texas	40% goal by 1994 (SB 1340, 1991)	No.	108
Utah	None.	No	19[9]
Vermont	50% by 2005 Mandate?	No	12 BC
Virginia	25% by 1995 (1989)	No	30[10]
Washington	50% goal by 1995 (Mandatory)	No	41[11]
West Virginia	50% by 2010 (1991)	Yes	2[4]
Wisconsin	40%	Yes	197[12]
Wyoming	None.	No	3

Another study by Biocycle on the State of Garbage in America identified the following information on state yard waste bans.   shows that 23 out of 50 states (or 46%) of U.S. states have yard waste bans in place.

Table 12 – MSW Landfill Disposal bans for Selected Materials

[9] Utah Department of Environmental Quality, 2008 Utah Compost Facility Inventory (Calendar 2007 Data), www.hazardouswaste.utah.gov/SWBranch/SWSection/PermittedSolidWasteLandfills.htm

[10] Mid Atlantic Composting Directory, Virginia Cooperative Extension, Publication 452-230, www.pubs.ext.vt.edu/452/452-230/452-230.html

[11] Washington Department of Ecology, 2007 Compost Facilities, http://www.ecy.wa.gov/programs/swfa/compost/

[12] Wisconsin Department of Natural Resources, Licensed Yard Waste Composting Facilities, www.dnr.state.wi.us/org/aw/wm/recycle/issues/compost.htm

State	*Yard Trimmings*	*Whole Tires*	*Used Oil*	*Lead-Acid Batteries*	*White Goods*	*Electronics*	*Others*
Alaska			x	x			
Arizona		x			x		
Arkansas	x	x		x		x[1]	
California		x	x		x	x	x[2]
Connecticut	x[3]			x			
Delaware	x[4]	x					
Florida		x	x	x	x		
Georgia	x[5]	x	x	x			
Idaho		x		x			
Illinois	x	x	x	x	x		x[6]
Indiana	x[7]	x		x			
Iowa	x	x	x	x	x		
Kansas		x					
Kentucky		x		x			
Louisiana		x	x	x	x		
Maine		x		x	x	x	x[8]
Maryland	x[9]	x	x		x		
Massachusetts	x	x		x	x	x	x[10]
Michigan	x	x	x	x			x[11]
Minnesota	x	x	x	x	x	x	
Mississippi		x		x			
Missouri	x	x	x	x	x		
Nebraska	x[12]	x	x	x	x		
New Hampshire	x	x		x		x	
New Jersey	x[13]						x[14]
New Mexico			x	x			
New York		x	x	x			
North Carolina	x	x	x	x	x		x[15]
North Dakota			x	x	x		
Ohio	x[9]	x					
Oregon		x	x	x	x		
Pennsylvania	x[16]	x		x			
Rhode Island[17]	x	x	x	x	x		x[18]
S. Carolina	x	x	x	x	x		
S. Dakota	x	x	x	x	x		
Tennessee		x	x	x			
Texas		x	x	x	x[19]		
Utah		x	x	x			
Vermont		x	x	x	x		x[20]
Virginia		x		x			
West Virginia[21]	x[22]	x	x	x			
Wisconsin	x	x	x	x	x	x[23]	x[18]
Wyoming				x			

[1]Regulations banning disposal of all computer and electronic equipment in landfills began January 1, 2008; [2]Fluorescent bulbs; [3]Grass clippings; [4]Yard waste ban at Cherry Island Landfill began in January 2008; [5]Yard trimmings are banned from MSW landfills designed and built to Subtitle D standards; Primary method of dealing with yard trimmings is disposal in inert and C&D landfills in state; [6]Potentially Infectious Medical Waste; [7]Leaves; also, woody vegetative matter greater than 3 feet in length is banned by statute; woody vegetative matter less than 3 ft in length is banned if it is not bagged, bundled, or otherwise contained; [8]Mercury containing products; [9]Separately collected yard waste is banned; [10]Glass, metal and plastic containers and recycled paper; Asphalt paving, brick, concrete, metal and wood were banned from disposal in July 2006; [11]Beverage containers; [12]Leaves and grass are banned April 1 through November 30; [13]Leaves only; [14]All other recyclable materials that any local government designates as a recyclable material; [15]Aluminum cans banned since 1994; beginning in 2009, wood pallets, plastic bottles and oil filters will be banned; Beginning in 2012, computers will no longer be accepted for disposal; [16]Truckloads comprised primarily of leaves; [17]All materials specified as mandatory recyclables are technically banned from landfill disposal; [18]Recyclable paper and containers; [19]With CFCs; [20]Oil based paint, mercury added products, ni-cad batteries; [21]Items checked are banned by state but individual landfills may have own rules; [22]West Virginia has a ban on yard waste; If no composting facility is available, landfills can usually get a waiver on the rule; [23]Generated by non-households.

5.2.1. Municipal Organic Waste

Table 13 contains a summary of key regulations related to diversion of organic waste from landfills to composting facilities.

Table 13 – Landfill Organic Waste Diversion Regulations

Regulation	Waste Applicability	Overview Summary / Goals	Implementation/ Enforcement
Mandatory Recycling and Composting - San Francisco Passed 6/9/2009 (San Francisco Supervisors, 2009)	Applies to all compostable waste generated within the City and County of San Francisco	100% segregation of trash, recyclables and compostable waste within the city. Specific requirements for multi-family and commercial properties, food/event managers, and haulers/processing facilities are established.	Specified containers must be provided at specific locations/events. Upon pickup, containers with contaminated material must be tagged with written notice following. Numerous tags/notices on the same container(s) result in administrative penalties for repeated violations not to exceed $100. Loads are available for inspection by the City.
Nova Scotia, effective 6/1/97 (Nova Scotia, 1996)	Compostable organic material - food waste, yard waste, soiled and non-recyclable paper	Nova Scotia is committed to achieving a national target of 50% waste diversion by the year 2000. Materials banned from landfill include beverage containers, corrugated cardboard, newsprint, scrap tires, used oil, lead-acid batteries, waste paint, automotive antifreeze, glass food containers, steel/tin cans, selected plastics and compostable organic materials.	Local municipalities given control over how to implement and enforce this regulation. Plans for each city must be provided to the Minister to ensure that the bans are implemented as described.
City of Seattle, WA table scrap recycling – effective April 2009 (City of Seattle, 2007 and Chan, 2007)	All single-family homes will be required to subscribe to food-waste recycling, a program that is now optional through the yard-waste collection program.	Supporting the City of Seattle's Zero Waste Strategy and to help meet its goal of diverting 72% of its garbage from the landfill by 2025, all single-family homes in Seattle must sign up for table-scrap recycling. Recycling food waste will be voluntary for apartments, as well as for businesses. A future ban of all organics from single family garbage will be considered once the collection system has been fully established.	Single family home residents are required to obtain new containers for food waste and pay for service. The city will not check whether they are actually dumping food in the new separate bin. Recycling food waste is voluntary for apartments, as well as for businesses. If a ban is implemented, it will likely follow a similar structure to existing bans: violators are fined or their garbage doesn't get picked up.
Massachusetts - Pending (Commonwealth of Massachusetts, 2006 - 310	Current landfill ban regulations apply to leaves, grass	The 2006 Solid Waste Master Plan states they will consider amending the waste regulations to add food waste to the list of materials	Under development; Under existing waste bans, no person is allowed to dispose or contract for disposal of

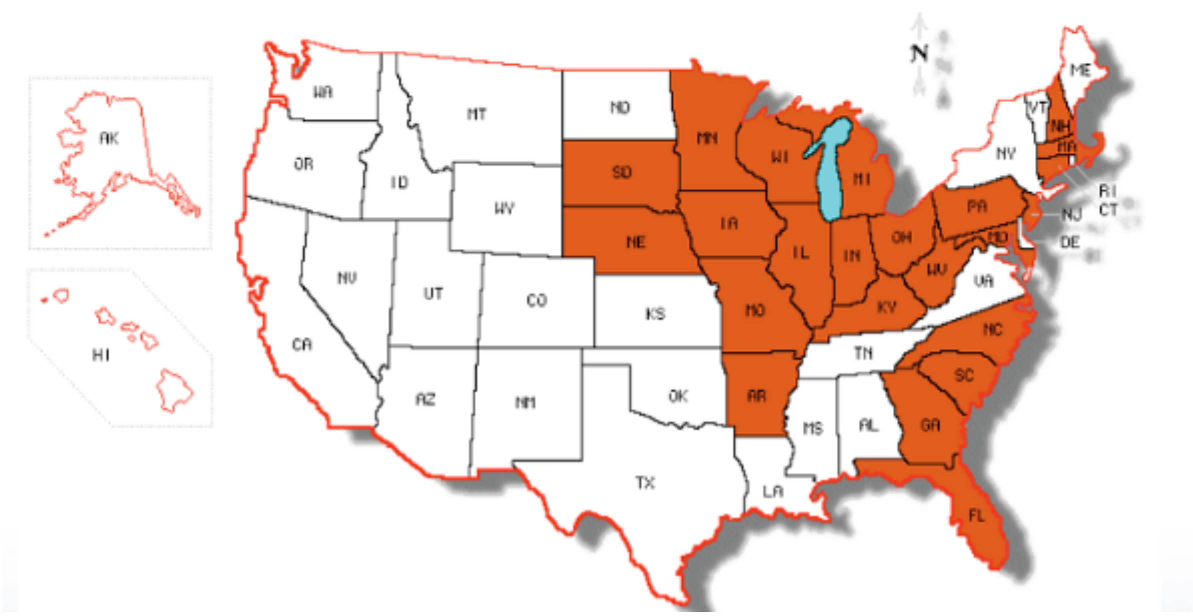
Regulation	Waste Applicability	Overview Summary / Goals	Implementation/ Enforcement
CMR 19.017, and Massachusetts Executive Office of Environmental Affairs. 2006)	clippings, weeds, hedge clippings, and brush up to 1 inch in diameter from disposal	banned from disposal once an adequate in-state food waste diversion infrastructure is in place. Targeted sectors include: residential, supermarkets, hospitals and other health care facilities, hotels and convention centers, colleges and universities, and state institutions such as prisons.	restricted materials. Where appropriate, Technical assistance and partnerships to stimulate market development are in place. Solid waste facilities, waste haulers and generators have a shared responsibility to comply with waste bans. MassDEP plans to conduct waste ban inspections at solid waste facilities. When haulers and generators of failed loads are identified, MassDEP will pursue enforcement against those entities.
Pennsylvania – (Preliminary review) (Hursh, 2007 and Pennsylvania Commodity Disposal Ban Review, 2008)	Source separated food waste only	Currently lack collection and management infrastructure to handle increased volume of food waste	Under development
Alameda County Ban on landfilling plant debris	All plant debris banned from landfill disposal within Alameda County (applies to two large landfills)	All plant debris	Jurisdictions required to prepare compliance plans

Food waste bans have only been implemented in a limited number of jurisdictions, but several other governments are contemplating adding mandatory food waste bans to existing landfill bans. While the methods and responsible agencies for implementation vary, most bans involve outreach and coordination with residences and businesses (as applicable), haulers, and the ability to perform waste audits to ensure compliance and identify areas for program reinforcement.

5.2.2. Yard Waste

As shown in Table 11 and Figure 7, almost half of the US States ban some form of yard trimmings from landfills. Other states have high recycling goals that perhaps serve a similar purpose (California, Washington). However, there is little data on the effectiveness of a given state's ban. It is fair to say that most of the state bans were put into effect as a means of preserving landfill capacity. Some states appear to have been more effective at implementing municipal and commercial composting programs. One study (DSM Environmental Services, 2004) makes the argument that states with bans have greater yard waste diversion, but each state tracks facilities and volumes differently enough to introduce some uncertainty at developing good, comparable per capita yard trimmings diversion numbers (also yard trimmings generation variances are not well understood-some states may generate more yard trimmings then others).

Figure 7 – States that Ban Yard Trimmings in Landfills



Source: Biocycle Magazine 'State of Garbage in America' (2008)

In recent years there have been several attempts to overturn state landfill bans on yard trimmings. For example, these have occurred in Iowa, Illinois, Nebraska, Missouri and Georgia. All of these to date with the exception of Nebraska (LB 776, 2006) have been unsuccessful.

With respect to the existing protocols available for composting GHG reduction projects, CCX, CDM and Alberta all allow yard waste as an accepted feedstock. CCX and CDM also allow biosolids as an accepted feedstock.

5.2.3. Biosolids

With respect to biosolids, in the U.S. the use and disposal of wastewater solids (sewage sludge) and biosolids is governed by US EPA regulations, 40 CFR Part 503. The Part 503 rule establishes risk-based standards for pollutants, pathogen and vector attraction reduction, and basic management requirements (e.g. agronomic loading rate). In order to land apply, biosolids are required to be treated to reach Class A or B pathogen reduction requirements. If they are treated to Class A they can be used without any restrictions. For Class B application is only allowed on agronomic crops (e.g. corn, wheat and soybeans) and a permit is required. In general, Class A is more expensive to achieve than Class B.

Even if treated to Class A standard, some regions have local bans on land application. In addition, there can be a great deal of public opposition to land application of biosolids. In the face of this, many municipalities opt to landfill biosolids. In certain states, landfilling can be less expensive than land application although typically the opposite is the case.

5.3. Distinguishing Additionality

5.3.1. How easy or hard might it be to distinguish between additional and non-additional projects?

Many States have a lengthy history of regulatory approaches to reduce the volume of materials sent to landfills. Some of the regulatory approaches are goal oriented, specifying only overall volume reductions that need to be made and leaving it to local jurisdictions how to implement the reductions. Because the goals do not typically target any specific waste stream, it does not appear that these would fall into the category of a regulatory requirement that would preclude inclusion in an offset protocol.

In contrast, a number of States have specific bans on materials sent to landfills. Most relevant in this discussion are the bans on yard waste. These are relevant to regulatory additionality, as the only materials that are not being diverted are essentially in non-compliance with regulations, even if the ban isn't well enforced.

In the case of biosolids, regulatory pressures sometimes dictate that the material be sent to a landfill instead of being landfarmed or applied to surface, which is the BAU practice. This study has not fully explored the various regional and local reasons that materials might be sent to landfills, although if the materials were deemed to be too hazardous for local land application, they would also seem unsuitable for composting operations.

Essentially, the distinguishing aspect for regulatory additionality is whether the material in question has been mandated to be diverted away from a landfill – e.g. yard waste and certain local bans aimed at food waste. Conversely, biosolids appear to represent a unique case where regulatory pressures and requirement sometimes dictate sending the material to a landfill.

6.0 BASELINE QUANTIFICATION

This section discusses the feasibility of quantifying baseline emission from diversion of organic waste from landfill to a composting project and seeks to answer the following questions:

- Is there sufficient data and information available to develop a standardized baseline for this project type?

6.1. Defining the Baseline

6.1.1. Decay Rate Constants

Anaerobic decomposition is a relatively complex microbial process that requires several different groups of organisms. It can be optimized or accelerated by managing moisture, pH, nutrient content, and temperature as well as by seeding with the necessary organisms. Decomposition rates in controlled digesters will be significantly faster and produce higher CH_4 yields that uncontrolled decomposition that occurs in sanitary landfills. Decomposition of feedstocks in controlled digesters is typically accomplished in less than 25 days.

There are many studies of decomposition of different feedstocks in digesters where most if not all of these factors listed above are controlled. This includes the study of simulated landfill decomposition by Eleazer et al. 1997. However, there are very limited studies of decomposition of specific feedstocks in landfills. There are studies (see discussion on landfill climate below) that include temperature and moisture measures within cells of sanitary landfills. There is limited information on time required for onset of methanogenisis (CH_4 production). There is also literature on settling in landfills, a phenomena that is related to decomposition and occurs in the early stages in the life of a landfill.

This suggests that there is a level of uncertainty involved in defining the rate of decomposition of specific feedstocks in a sanitary landfill. This uncertainty can be dealt with in two different ways. Results from controlled anaerobic digestions studies of specific feedstocks can be used as a baseline. Rates of decomposition as well as total methane yield can be altered to reflect the relative inefficiency of decomposition in a landfill. An 'inefficiency factor' can be applied to adapt the results from controlled digestion studies to the retarded decomposition that occurs in a landfill.

One example of this would be decomposition and methane yield of food scraps. Rates of decomposition in controlled studies range from 10-120 days with a CO_2 equiv CH_4 yield of 4.25 -8 Mg per dry Mg of food scraps (Brown et al., 2008). Studies of gas generation in landfills measured the onset of CH_4 formation at d 20, 30 or d 70. Settling is shown to occur within the first year of waste deposition. This suggests that an 8 x[13] inefficiency factor could

[13] The 8 x rate was determined by considering the onset of methane generation in landfill cells in combination with reported rates of settling and comparing these to time required for methane generation in controlled anaerobic digesters. It was assumed that food waste would be the most readily degraded of the different components of MSW and so would be likely to be one of the first substrates to emit methane in a landfill cell. In anaerobic digestion studies, CH4 formation from food waste decomposition is generally exhausted from between d 10-110. Applying a factor of 8 to this would lead to decomposition of food waste post placement in a landfill cell from d 80 to d 880. This is well within the range of the onset of methanogenisis and initial settling related to decomposition in landfills.

conservatively be applied for decomposition rate. This would increase the decomposition time for food scraps from 10-120 days to 80 to 960 days. This would be in agreement with both rates of methane formation in landfills as well as settling rates.

Another approach would be to set decay rate constants for different waste materials and only include the most rapidly decaying materials in the protocol. This was the approach taken by the CDM and the CCX protocols. Whereas the CDM protocol had different decay rate constants based on the external climate where the landfill was situated, the CCX protocol used single rates across all landfills based on studies that suggest that the climate within a landfill is generally controlled by waste decomposition rather than external factors. The problem with this approach is the overreliance on a single study for k values. A potential alternative would be to use the decay rate constants for specific chemical components of waste included in the Meima et al (2008) study to develop decay rate constants for specific categories of wastes. As this study uses a standard engineering model to predict settling, it is also a conservative indicator of potential decay rate constants.

6.1.2. Gas Collection Efficiency Baseline

Landfill gas collection efficiency is a highly contentious topic. The US EPA uses a baseline efficiency of 75% while the IPCC uses a range of 40-50%. A study of landfills in California compared predicted and actual gas generation across 35 landfills (Themelis and Ulloa, (2007). Efficiency (actual gas collected/predicted gas produced) ranged from 6 to 100 percent with an average efficiency of 35%. Efficiency in gas collection will vary by landfill, section of landfill (active or closed) and season. A straightforward way to avoid this uncertainty is to focus the protocol on the period of the landfill cell where no gas collection systems are in operation. This restricts the protocol to highly putrescible materials and reduces uncertainty on gas collection efficiency. This was the approach used by the CCX protocol.

6.2. Standard Methods for Quantifying Methane from Organic Waste in Landfills

6.2.1. Landfill Decay Rates

Anaerobic digestion studies have shown rapid degradation and high CH_4 yields from of a range of feedstocks (e.g. Zhang et al., 2006, Hansen et al., 2004). Digestion in these studies is likely accelerated in comparison to anaerobic digestion in landfills due to a number of factors. Anaerobic digesters generally operate at optimal moisture and temperature for decomposition. Wet digestion generally occurs at >90% H2O and at temperatures in the mesophilic (35°C) or thermophilic (55°C) range (Metcalf et al., 2003). Decomposition is also accelerated by seeding reactors with bacterial innoculum (Davidson et al., 2007). In most cases, material is ground or homogenized prior to digestion. This is done to both ensure that representative samples are tested and to accommodate the requirements of wet digesters. Grinding materials also reduces particle size, which in turn accelerates the rate of digestion.

Anaerobic degradation rates of specific materials in landfills have not been documented in situ. In a study that was conducted to characterize the anaerobic degradability of different components of MSW, CH_4 yields of different materials were measured along with the time required for decomposition (Eleazer et al., 1997). The values from this study have been used to calculate potential decay rates of the different components of MSW (Barlaz, personal communication). Those decay rates are shown in Table 14 below.

Table 14 – Landfill Decay Rates

Component	Conventional Landfill Decay Rate (K rate units)	Bioreactor Landfill Decay Rate (K rate units)
MSW	0.04	0.1
Leaves	0.25	0.63
Grass	0.31	0.77
Branches	0.12	0.30
Old News Print	0.04	0.11
Old Corrugated Containers	0.03	0.08
Magazines	0.16	0.40
Food Waste	0.11	0.28

A concern with using these decay rates as a measure of actual degradation rates in a landfill is that the study that provided the data for these rates used similar methods to digestion studies and so is likely not representative of the behavior of materials in an actual landfill. In the Eleazer et al (1997) study, moisture was maintained at optimal levels for decomposition, materials were shredded prior to incubation, reactors were seeded with microbial inoculum, temperatures were maintained at 40°C, leachate was re-circulated after pH was neutralized, and nutrients were added to meet target levels. This type of experimental set up is likely to generate accelerated rates of decomposition for feedstocks that are dry and nutrient poor such as old news print, old corrugated containers and magazines. For example, in the Hansen et al (2004) study only 63% of the calculated CH_4 potential for brown paper bags was generated (compared to approximately 80-90% for other waste such as food waste). In this study, an inoculum was used, moisture was added, pH was regulated and materials were shredded prior to digestion. However, no supplemental nutrients were added (which would assist with decomposition).

In another study conducted to test the efficacy of anaerobic decomposition relative to aerobic decomposition Erses et al. (2007) measured rates of decomposition and gas generation in lab scale reactors maintained at 32°C and at optimal moisture. The waste material used in this study consisted of 45% food and garden scraps and 15% paper. No inocula were added to the reactors and pH was not controlled. Methane generation in the anaerobic cell commenced at approximately day 200 with CH_4 concentrations approaching 50% of total gas volume at d 350. The pH in the cell increased at about d 400 from < 6 to > 7. These results indicate that anaerobic decomposition was potentially delayed due to low microbial populations and a build up of acids.

Another study was conducted to determine the efficacy of alternate inocula on anaerobic decomposition of the organic fraction of MSW. Here, the author's note that in many countries that are considering use of AD, appropriate inocula is not available and start up times for reactors can be up to 1 year (Maroun and El Fadel, 2007). Using horse manure, the authors were able to achieve relatively stable digestion with a hydraulic retention time of 148 days. This is in comparison to the approximately 15 day retention times in studies where appropriate inocula has been supplied (e.g. Grey, 2008; Zhang et al., 2006; Hansen et al., 2004).

It may also be possible to estimate the relative efficiency of landfill degradation in comparison to anaerobic digesters using studies that measure conditions in landfills. One study noted that methane production was observed when % H2O was as low as 10%. Field capacity or incipient free moisture conditions occurred at 40% moisture (Hartz and Ham, 1983). Leachate recirculation significantly increased methane generation in comparison to baseline conditions. Similar results were found by Bogner (1990).

Temperature is another factor that will influence the rate of degradation. The Clean Development Mechanism protocol for methane avoidance for landfill diversion to composting operations uses different k rates (units for decay rates) for landfills located in different climates. The Chicago Climate Exchange protocol uses a single decay rate constant for each waste type based on the assumption that conditions within sanitary landfills will be similar across different climate regimes.

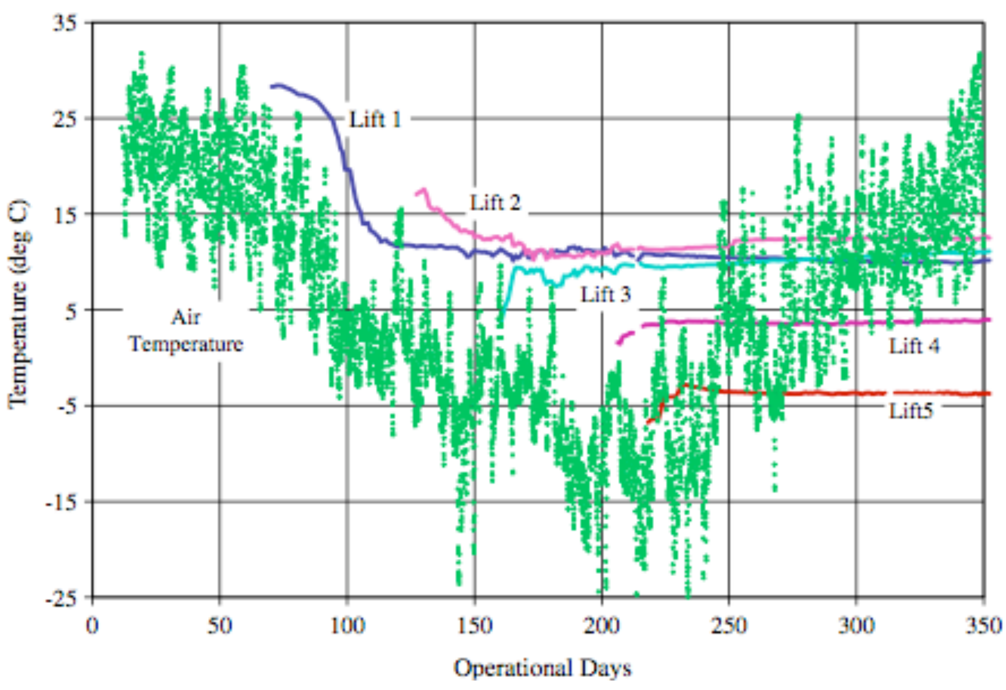
In one study, temperature and gas composition were measured in a sanitary landfill cell in the south of France. The cell was filled for approximately 1 year, beginning in December with a total volume of 200 000 m^3 before it was covered with a 30 cm ground layer and an airtight membrane. The cell contained 63.4% municipal waste with an additional 1.8% sewage sludge and oil and 1.1% fruit and vegetables. During filling, waste was deposited in the cell weekly at depths ranging from 0.2-0.6 m. The bottom and sides of the cell were constructed of compacted clay. Probes were placed at different depths within the cell and temperature, moisture and gas composition were measured. Temperature at the surface 4.5 m layer of waste increased from about 25 °C to 40°C within 20 days and was consistently higher than ambient temperature. The temperature remained steady for 160 days of measurements with the final cover being put into place at day 55. Methane was detected at this surface layer on day 20 and CH_4 concentration stabilized at about day 100 at about 50% of gas composition. Methane was consistently detected at this level for the deeper probes used in the study. Sample moisture varied from 20-50%.

Another study measured temperature and moisture in different landfills in Germany (Bäumler and Kögel-Knabner, 2008). Waste of different ages, as well as bioreactor and non bioreactor landfills were included in this survey. Sampled landfills were constructed beginning in 1977, 1978, and 1979 with final covers placed between 1999 and 2001. Temperatures in the surface 2 m of the waste were < 20°C due to the influence of climatic conditions at the landfill sites. However, temperatures increased to 50°C with depth particularly in the younger materials. Samples at older sites showed a decline in temperature to < 30°C which the authors attribute to the exhaustion of readily degradable feedstocks. Moisture varied from 60 to 420 g kg-1 (generally between 100 and 400 g kg-1) independent of age and depth of the waste materials. The authors suggest that moisture content was controlled by the structure and bedding of the waste when it decomposed resulting in preferential flow. The authors observed a ratio of cellulose: non- cellulosic carbohydrates > 1 indicating the dominance of paper waste and its resistance to decomposition.

Another study measured temperatures of geomembranes located at the base and surface of landfill cells for two cells in the same landfill (Koerner and Koerner, 2006). One cell was maintained as a dry cell and the other as a wet cell. Temperatures at the surface of both cells ranged from 0-30°C and reflect seasonal variations. The temperatures of the membranes at the base of the dry cell averaged 20°C for 5.5 years and then increased to 30°C. In the wet cell the temperature started at 25°C and increased to 41-46°C during the 3.7 years of monitoring. These results are in agreement with Gholamifard et al., 2008. Here temperatures were measured in different cells and at different depths in a landfill in France. Temperature increased with depth averaging > 40°C at 3 m for March. Ambient temperature was not provided but is likely significantly lower.

The only study that suggests a greater and prolonged influence of ambient temperature on landfill temperature used an experimental cell in Michigan. Zhao et al., (2008) monitored temperature, moisture, and gas generation in a bioreactor landfill cell that contained approximately 32,400 Mg of waste. Initial moisture content of the waste was 25%. Average moisture reached 37% with leachate recirculation and stabilized at 60% for all depths but the surface depth after 200 days. The CH_4 concentration of the initial lift was 25% at day 70, increased and stabilized at 50% by day 200. Methane concentration for the other lifts remained low for the duration of the study. The authors attribute this to cooler temperatures in the cells that were filled during the winter months. Temperatures in all cells were clearly influenced by ambient temperatures. Ambient temperature and the temperature in different depths of the cell are shown in Figure 8 below.

Figure 8 – Temperature Changes in Landfill at Different Depths



A comparison of the findings of these studies is shown in Table 15 below.

Table 15 – Comparison of Landfill Characteristics identified by Three Studies

Study	Waste quantity	Temperature	% moisture	CH_4 onset

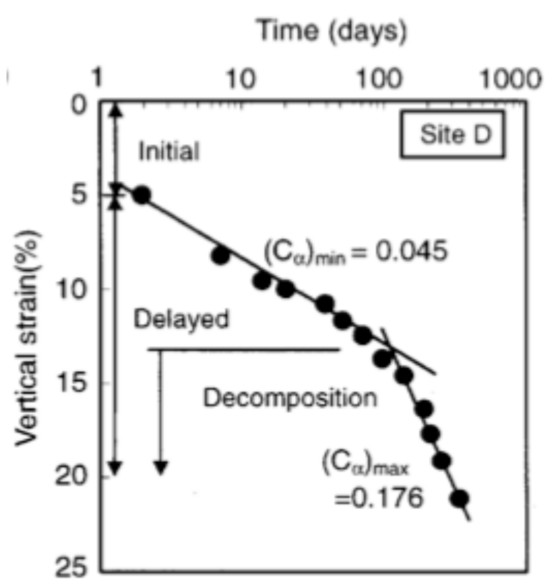
Study	Waste quantity	Temperature	% moisture	CH_4 onset
Levebre et al, 2000	200 000	40 °C	20-50%	Day 20
Bäumler and Kögel-Knabner, 2008	893 000	20° C at surface 50 °C at depth	.6-42%, primarily 10-35%	
Zhao et al., 2008	32 400	ambient	25-60	Day 70

The results from these studies suggest that if a landfill is of sufficient size, internal temperature conditions will be controlled by decomposition of the waste materials and will generally approach the standard temperatures for mesophilic digestion.

6.2.2. Landfill Settling

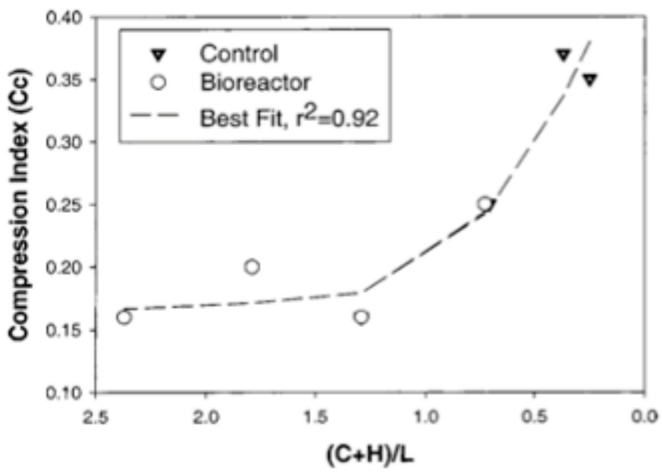
In sanitary landfills waste is deposited into a cell and compacted. Despite compaction on placement into the cell, landfills regularly experience settling during the first years after filling (Hossain et al., 2003; Ling et al., 1998; Park et al., 2002). Settlement as a result of decomposition can reduce the thickness of MSW by 18-24% and is potentially more significant than volume reduction that results from placing additional loads on the landfill surface (Park et al., 2002). Park et al (2002) divides settlement into two initial phases; one due to compression from placement of additional waste and the second due to the rapid decomposition of organics. A model of the compression and decomposition phases is shown in Figure 9 below.

Figure 9 – Landfill Compression Phases



Hossain et al. (2003) tested samples representative of residential MSW for compressibility in lab scale reactors. Reactors were maintained either with or without leachate recirculation. They quantified the state of decomposition using both CH_4 yield and the ratio of cellulose plus hemicellulose to lignin as indicators of decomposition. Compressibility increased over time and was correlated to decomposition as measured by the change in this ratio. During the course of the study this ratio decreased from 1.29 to 0.25. Methane production increased rapidly in the bioreactor units from day 30 to day 60. In the control units, methane production began at D 90 and was much less pronounced. Samples were collected for150 days. A graph of the compression index and the cellulose to lignin ratio is shown in Figure 10 below.

Figure 10 – Landfill Compression Cellulose to Lignin Ratio



The observed increase in compressibility and settlement of landfills suggest the rapid decomposition of certain organic fractions of MSW. The studies on the environment in landfills and settling indicate that temperatures within the landfill cell are conducive to anaerobic degradation. Moisture content in landfills appears to be variable with increased moisture and accelerated decomposition in bioreactor landfills. Settling during the initial period of the life of a cell appears to be representative of decomposition of material within the cell. The next question is which components of the organic fraction of MSW are responsible for the initial CH_4 release and settling.

6.2.3. Decomposition of Municipal Solid Waste

Authors have taken different approaches to determine which materials decompose most rapidly within the cells. Hossain et al (2003) focused on the ratio of hemicellulose and cellulose to lignin as an indication of decomposition. Decreases in this ratio were taken to represent breakdown of cellulose. A similar approach was taken in a review of decomposition of wood products by Barlaz (2008), who notes that cellulose, hemicellulose and lignin are the primary organic compounds in landfilled organics. In one landfill in Kentucky, the average cellulose, hemicellulose and lignin concentrations of 87 1-3 year old refuse samples was 36.7, 8.2 and 18.5%, respectively. In another study of 7-21 year old waste excavated from a landfill in Berkeley, Wang et al. (1994) found cellulose concentrations ranging from 0.9% to 11.7% with associated lignin concentrations of 85.6 and 70.9%. A significant decrease in the ratio of cellulose + hemicellulose: lignin is taken to mean both that lignin persists in a landfill environment and that cellulose decomposes in a landfill environment. In another study, waste from a number of landfills in Germany was analyzed for chemical composition ((Bäumler and Kögel-Knabner, 2008). The authors found high concentrations of cellulose in waste material of different ages and interpreted this to mean that paper waste does not readily decompose in a landfill.

Eleazer et al. (1997) notes that food waste is composed primarily of cellulose (55.4%), hemicellulose (7.2%) and lignin (11.4%). This is questionable based on the chemical content of different types of food (USDA ARS) as well as the fact that cellulose and lignin are not digestible by humans. Cellulose is commonly found in wood, cotton and paper. Using this measure as an indication of decomposition removes a majority of food waste and a large portion of fresh plant material from consideration.

Meima et al. (2008) noted that recent models have been developed to predict gas generation and decomposition in landfills using Monod kinetics. Monod kinetics is a means to calculate decomposition of a particular substrate when the bacteria responsible for degrading the substrate are not present at a constant population. It takes into account the growth rate of the bacterial population in determining the decay rate of a particular substrate. Standard models using Monod kinetics require input on over 40 parameters for use. Meima et al (2008) attempt to simplify the model by reducing the number of input parameters. In addition to factors on the landfill environment (water content, temperature, and pH), the authors provide a range of decay rates for different components of the waste materials. These include rates for first order hydrolysis (the first step in the anaerobic decomposition reaction) for carbohydrates and proteins as well as growth rates for anaerobic biomass on glucose, propionate, acetate and butyrate. The authors conclude that landfill environment and the presence of readily decomposable substrates are the primary parameters controlling the rate of CH_4 production.

Sormunen et al. (2008) sampled wastes of different ages from two landfills in Finland. One landfill had been operating for 17 years and the other for 48 years. The younger landfill had a higher ratio of VS:TS. Biological CH_4 potential (BMP) and the ratio of VS:TS was highest in the middle and top layers. The % of wood was similar across all depths. The authors noted that the proportion of paper and cardboard was lowest at the bottom depth indicating that the material had decomposed. There was no detection of food related residuals at any depth. Ximenes et al. (2008) sampled landfills in Sydney, Australia and also found that wood products persist in a landfill environment. In this study different types of wood from three landfills that had been closed for 19, 29 and 46 years were sampled. Moisture content of the wood ranged from 41.6-66.8%. The total carbon, cellulose, hemicellulose and lignin concentrations of specific wood types from the landfills were measured and compared to fresh samples of the same species. There was no evidence of decomposition in the two younger landfills. In the oldest landfill sampled up to 18% of the original carbon content of the wood had decomposed.

6.2.4. Nitrous Oxide

Emissions of N_2O from landfills is mentioned in the IPCC guidance document on waste, however no specific information is given on sources of N_2O from different components of MSW or on quantification of N_2O emissions (Pipatti et al., 2006). There are some studies that are pertinent to this topic. Börjesson and Svensson (1997) measured N_2O emissions from landfills in Sweden where soil or municipal biosolids were used for cover soil. Here biosolids were applied as cover soil at a depth of 0.5 to 1 m. Nitrous oxide emissions from the mineral soil ranged from -0.0017 to 1.07 mg N_2O-N/m^2/h. Emissions from the areas covered with biosolids ranged from -0.011 to 35.7 mg N_2O-N/m^2/h.

Other studies have documented N_2O emissions from landfills. Rinne et al. (2005) measured N_2O from the surface of a landfill in Finland using two different techniques. Waste at the landfill sampled for this study was compacted immediately and covered daily with a 20 cm layer of soil. The soil used for cover was a mixture of organic soil (20-40% organic matter) and mineral soil. Gases at the landfill were partially recovered and flared. Measurements for this study were made from the active face of the landfill. Gas was measured using an enclosure technique as well as an eddy covariance method. The first is the more standard approach but can lead to error as conditions inside the measurement chamber may not reflect actual site conditions. Eddy covariance measures are taken above the soil surface and are therefore less likely to alter the environment. Nitrous oxide emissions from the landfill surface were 2.7 mg N m-2h-1 and 6.0 mg N m-2h-1 for the eddy covariance and enclosure methods, respectively. These values are approximately 1 order of magnitude higher than the highest emissions reported from agricultural soils in Northern Europe.

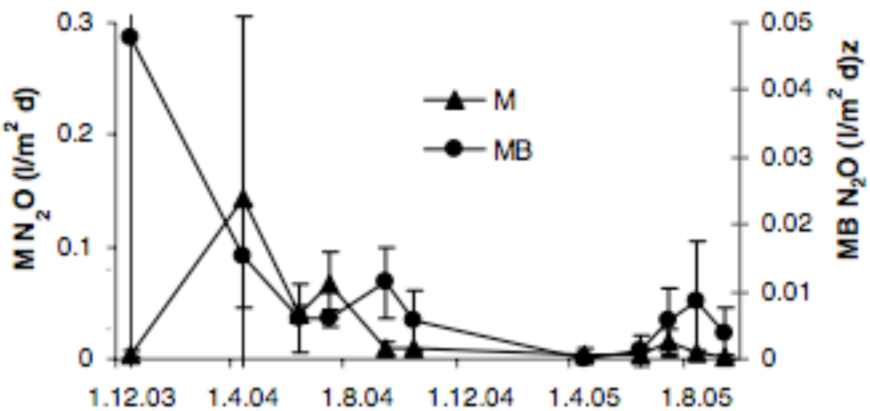
Zhang et al. (2009) measured N_2O and CH_4 fluxes from three landfills in China seasonally from the fall of 2006 through the winter of 2008. Two of the sites had gas collection systems and high clay cover soils (45 and 39 % clay, 1 and 2% sand, for sites A and B respectively). The other site had no gas collection system and a sandy cover soil (22 % clay and 24% sand for site C). Emissions of N_2O from the 3 sites ranged from:

- Site A 0.133 to 0.725 mg N_2O-N $m^{-2}h^{-1}$
- Site B -0.036 to 2.483 mg N_2O-N $m^{-2}h^{-1}$
- Site C -0.102 to 0.52 mg N_2O-N $m^{-2}h^{-1}$

The three studies of N_2O emissions from landfills report values that are generally within the same range. Values for these studies were collected from landfills in China, Finland and Sweden. The consistency across sites indicates a high potential for fairly consistent fluxes. These values are expressed as emissions per m-2 rather than per Mg of MSW. They also don't reflect the N_2O generation potential of individual components of MSW. The values from Börjesson and Svensson (1997) for the landfills covered with biosolids can be used to estimate the N_2O generation potential for a mono-fill. Assuming a depth of cover soil of 0.5 m and a moisture content of 25%, the dry weight of biosolids per m^2 would be 0.125 Mg. The emissions from the biosolids covered portion of the landfill ranged from 6 to 35 x higher than from the portion covered with the mineral soil. Assuming an emissions rate of 10 mg N_2O-N m-2h-1, the CO_2 equivalent emissions per dry Mg of biosolids over a 24 hour period would be 0.58 kg CO_2. For a one year period this would be 0.210 Mg CO_2 per dry Mg biosolids.

One other study can be used to inform estimates of the N_2O emissions potential of landfilled MSW. Sormunen et al. (2008b) compared CH_4 and N_2O emissions from mechanically and mechanically-biologically treated municipal solid waste collected from an area of Finland with an active recycling and organics diversion program. In this area, biowaste and papers are separated in buildings with more than 5 households. Additionally, biowaste, paper and cardboard are collected separately where the total waste stream > 20 kg per week. The residual material is then treated to remove metals and shredded and screened. This residual fraction was then placed in large scale lysimeters for the mechanical treatment (M) or composted in aerated tunnels followed by passive aeration using wood chips or the larger size fraction as support material for the mechanical- biological treatment (MB). Nitrous oxide emissions were one order of magnitude higher for the M treated materials in comparison to the MB treated materials. The data from the study is shown in Figure 11 below.

Figure 11 – Nitrous Oxide Emissions from Mechanical versus Mechanical-Biological Treatment



The limited number of studies of emissions of N_2O from landfills demonstrate that landfills are a source of N_2O. The microbial pathways conditions for N_2O formation also suggest that landfills provide a suitable environment for the formation of this gas, both in oxidation of ammonia by methanotrophic bacteria and in denitrification reactions.

The Clean Development Mechanism protocol for methane avoidance for landfill diversion to composting operations takes a default debit for fugitive N_2O emissions during the composting process (Clean Development Mechanism, 2008). However, the anoxic conditions required for N_2O formation are potentially likely to occur in a landfill environment.

6.2.5. Landfill CH_4 Oxidization

The US EPA estimates a 75% gas capture efficiency across the lifespan of landfills with gas collection systems (US EPA 1998, 1999). However, this assumption does not take into account differences in emissions for different stages in the operation of a landfill. Rates of CH_4 emissions are likely to vary by cover soil, management, local climate, waste composition and compaction ratio and portion of the landfill in question (Mosher et al., 1999; Chanton et al., 2009). Emissions will be greatest in the active portion of the landfill before final cover has been placed. It is relatively straight forward to assign a 0% gas collection efficiency to landfills or portions of landfills where no gas collection system is in place. In these cases it is also likely that the cover material will oxidize a portion of the CH_4 that is produced. The CDM and CCX protocols for landfill diversion to compost facilities use a 10% default value for the percentage of CH_4 that is oxidized by the soil or daily cover. A recent review of research on this topic suggests that a 25% value for CH_4 oxidation is more appropriate for final cover materials at landfills (Chanton et al., 2009). This review did not distinguish between oxidation of CH_4 emissions from the active portion of the landfill versus final cover. It is likely that emissions from daily cover will vary based on the type of cover material used.

6.3. GHG Potential of Individual Compost Feedstocks In Landfill

6.3.1. Food Waste

Food scraps can be divided into different categories using the USDA food pyramid as a guideline. The food pyramid divides foods into the following categories

- Grains
- Vegetables, dry beans and peas
- Fruits
- Milk Group
- Meat and bean group
- Oils
- Solid fats and sugars

The chemical composition of a range of foods is available from the USDA (http://www.nal.usda.gov/fnic/foodcomp/search/). The basic chemical content of select examples from each of the food pyramid groups is shown in Table 16 below.

Table 16 – Chemical Content of Different Food Wastes

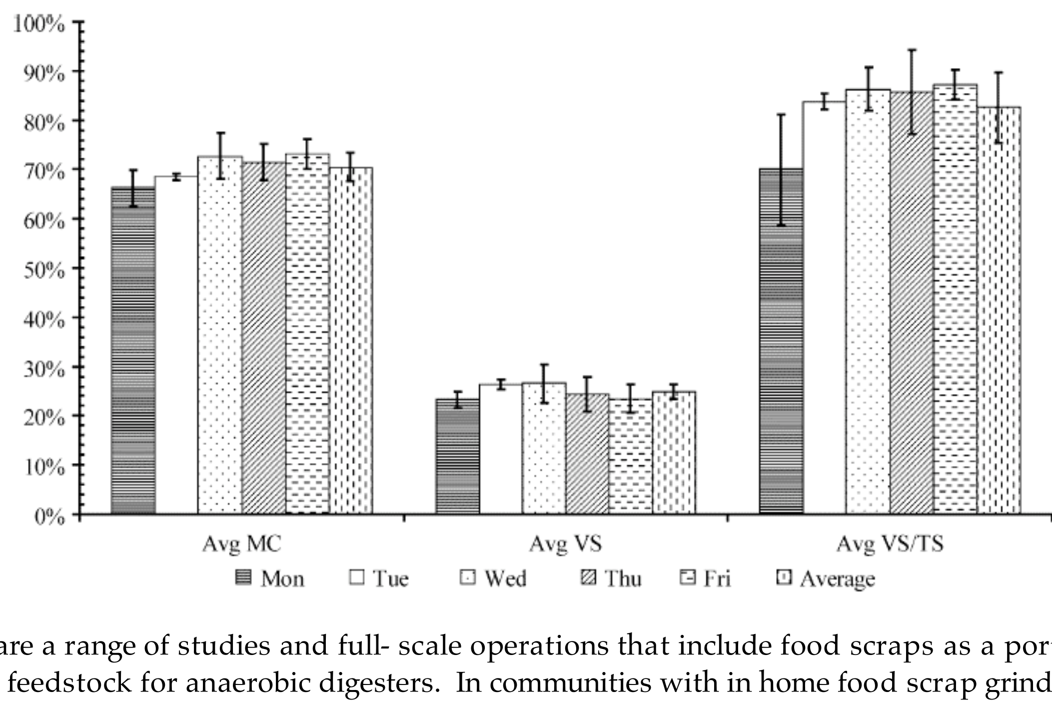
Food group	Water	Carbohydrate	Protein	Fat	Energy
		%			kcal per 100 g
Fruit					
Apples	86	13.8	0.26	0.19	52
Vegetables					
Broccoli raw	90.7	5.25	2.98	0.35	28
Cereal					
Rice (cooked)	73	23.5	2.32	0.83	112
Meat					
Pork (cooked)	51.8	0	21.9	25.4	323
Dairy					
Cheddar cheese	36.75	1.28	24.9	33.1	403
Fats					
Olive oil	0	0	0	100	884
Sweets					
Almond Joy	8.2	59.5	4.1	26.9	479
Prepared foods					
Stouffers salisbury steak in gravy + macaroni and cheese	80	7.47	6.4	5.2	102

In general, the primary constituent of food scraps is water. Other compounds commonly found in food scraps include carbohydrates, proteins, and fats. These compounds are readily decomposed under anaerobic conditions. Food waste collected from commercial or domestic sources will contain a mixture of these different food groups and may vary in composition by a number of factors including region of the Country and time of year. Different studies have characterized food waste from different sources in Korea, Germany and India. Moisture content in these studies varies from 74-90%, the ratio of volatile solids/total solids ranges from 80 to 97% and the C:N ratio varies from 14.7 to 36.4 (Zhang et al., 2006).

A study of source sorted municipal organic waste from cities in Denmark used an analysis of variance to test the importance of dwelling type, season and collection system on waste composition (Hansen et al., 2007). Fat and protein content were not affected by any of these parameters. The calorific value of the material ranged from 19.7 to 20.8 Mj/kg organic fraction of dry matter. Plastic contamination was highest in the two cities that collected from the smallest number of households and lowest in the city that collected from the 2nd highest number of households.

In a study of anaerobic digestion potential of food waste in the United States, food scraps were collected from 500 sources including markets, restaurants, hotels and businesses in the San Francisco Bay area (Zhang et al., 2006). Average moisture content, volatile solids and ratio of volatile solids/total solids from the different collections are shown in Figure 12 below.

Figure 12 – Food Waste Characteristics



There are a range of studies and full- scale operations that include food scraps as a portion of or sole feedstock for anaerobic digesters. In communities with in home food scrap grinders, food waste is digested as part of the traditional wastewater treatment plant influent. Co-digestion and mono-digestion of food scraps has been tested on a full scale basis at the East Bay MUD wastewater treatment facility in Oakland, CA (Gray, 2008).

Methane production in mono digestion of food waste averaged 420 m^3 CH_4 per dry metric ton with a range of 300-530. In comparison, traditional wastewater solids produced an average of 310 m^3 CH_4 per dry metric ton with a range of 230-390. This was based on a 15 day solids retention time in the digester. Volatile solids reduction averaged 73.8% when the digester was run at a mesophilic temperature range and 80.8 when it was run under a thermophilic temperature range. Another study found a VS degradation rate of about 80% with a CH_4 yield of 300-400 Nm^3 CH_4/ton with a retention time of 15 days (Davidsson et al., 2007).

These results suggest that mixed food waste readily decomposes under anaerobic conditions with relatively predictable CH_4 yield and associated VS destruction and minor variation in composition. Another report from this group tested the variability of characterization of source separated household organics based on sampling procedures (la Cour Jansen et al, 2004). Variables tested included ash content, crude fibers, crude fat and protein, sugar, starch, enzyme –digestible organic matter and calorific value. Different methods of sub-sampling included shredding, mixing, blending, drying and milling and high speed blending were tested on subsets of samples tested over the course of a year. The variability related to sample collection was similar to the variability of the samples over time, indicating that different methods for sample collection will not skew analysis of substrate characteristics. In addition, variability over time was low.

6.3.2. Other Feedstocks

A range of other organic feedstocks sent to compost facilities are also likely to decompose under anaerobic conditions. For example, different grass species were tested for biogas generation potential with rates of VS destruction ranging from 65-86% (Mähnert et al., 2005). Feedstocks for current and planned anaerobic digestion projects are listed below

Feedstocks for anaerobic digestion (from Dennis Totzke, Biocycle, 2009) are outlined below.

- Aircraft deicing fluid
- Beet pulp
- Brewery waste
- Cheese whey
- Chicken manure
- Clarifier skimmings
- DAF float
- Processed algae
- Fermentation Wastes
- Glycerin
- Grass clippings
- Grease trap pump out
- Meat processing wastes
- Organic fraction of MSW
- Outdated beverages
- Outdated food products
- Restaurant wastes
- Snack food waste
- Thin silage
- Various spent grains
- Vegetable wastes

This suggests that a wide range of substrates are likely to decompose under anaerobic conditions. Several studies have attempted to quantify CH_4 generation potential based on chemical composition and VS content of different feedstocks (Davidsson et al., 2007; Eleazer et al., 1997; Hansen et al., 2004; Wang et al., 1997).

Volatile Solids

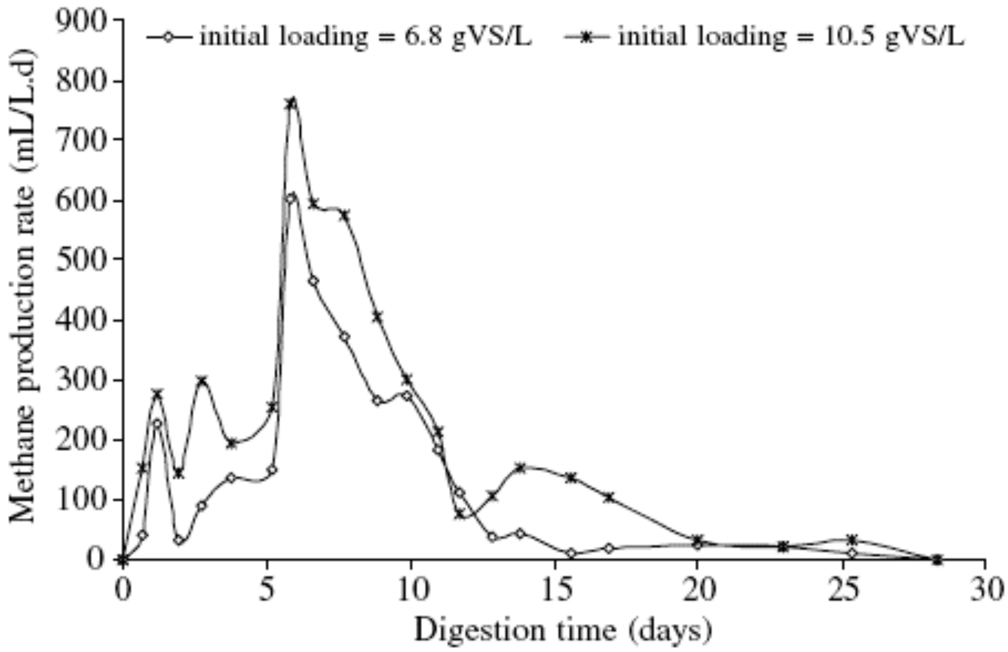
The volatile solids content of a substrate can be used to predict the methane generation potential of a material. Volatile solids is straight- forward to measure, does not require specialized equipment or a long turn around time. A summary of the EPA method for volatile solids measures is as follows (US EPA, 2001):

- Sample aliquot of 25-50 g dried at 103-105° C to evaporate water

- Residue is cooled, weighed and dried at 550 ° C to drive off volatile solids
- Volatile solids are calculated as the difference in the dried sample weight prior to and after heating at 550 ° C

Anaerobic digestion provides optimal conditions for VS destruction and CH_4 production. A typical pattern of methane release for batch fed digesters is shown in Figure 13 below (Zhang et al., 2006).

Figure 13 – Methane Release for Batch Fed Digesters



In digesters, CH_4 production will peak before all volatile solids are destroyed. An engineering text uses the equation in Figure 14 to predict VS destruction in wastewater treatment plants (Metcalf et al., 2003):

Figure 14 – Volatile Solids Destruction in Wastewater Treatment Plants

$$V_d = 13.7 \ln(\text{solids retention time[days]}) + 18.9$$

Using this equation, 82% of VS will be destroyed with a solids retention time of 100 days. The quantity of VS destroyed and associated CH_4 production during anaerobic digestion or decomposition will vary by substrate. An extension publication for anaerobic digestion of different animal manures predicted a range of potential VS destruction from 45% (beef cattle) to 60% (poultry layers and broilers)(Hansen, 2004). Volatile solids destruction in food waste digesters has been reported to be between 77 and 89% (Zhang et al., 2006).

For certain feedstocks, including animal manures, municipal wastewater biosolids, food scraps and grass, there is sufficient information available on VS destruction potential under anaerobic conditions that VS can be used as a measure of CH_4 generation potential. For other, less well characterized feedstocks (examples include paper wastes), controlled digestion studies would be required to determine a reasonable approximation of potential VS destruction. Once this baseline has been established, VS can be used to measure CH_4 generation potential.

7.0 POTENTIAL REDUCTION OPPORTUNITY AND COST OF REDUCTIONS

The objectives of this section were to

- Would it be feasible and accurate to apply standardized methods for estimating project emissions?
- assess the technical and economically feasible reduction potential for project activities within the United States.
- estimate the typical cost of achieving reductions for potential project activities ($/ton of CO_2-equivalent).

7.1. Project Emissions Quantification Feasibility

7.1.1. Would it be feasible and accurate to apply standardized methods for estimating project emissions?

There are several approaches that could be used to establish project emissions for composting operations. There are two components of composting that need to be considered, energy use during composting and fugitive emissions from compost feedstocks. For energy use during composting, it is possible to track fuel and electrical use at each compost site. Energy use estimated for different types of composting systems have also been published (Brown et al., 2008). While certain types of systems are less energy intensive than others (i.e. windrow versus enclosed systems), energy use is minimal in comparison to CH_4 avoidance. It is also true that energy use will likely be similar for landfill operations as for composting operations. These factors suggest that for the vast majority of projects energy use during composting can be considered de minimis. This was the approach taken by the CCX protocol, although in that protocol there is a provision to calculate energy use. If it is below a particular threshold, it is discounted.

The second factor to be considered is fugitive emissions during composting. Here, both the Alberta and the CDM protocols take default emissions deductions from potential credits for potential release of N_2O and CH_4 from composting operations. The CDM protocol takes a debit of 0.043 kg N_2O per tonne of wet compost assuming 65% solids. This is equivalent to 0.043 kg per tonne/1000 kg per tonne or 0.00004 kg N_2O per kg compost. Methane emissions for the CDM protocol are calculated based on oxygen measures taken on a preapproved sampling scheme. The Alberta protocol deducts 0.004 kg CH_4 per kg compost and 0.0003 kg N_2O per kg compost. It is not clear from the text of the protocol if this is on a dry or wet weight basis and if this is on the finished compost or on initial compost piles.

There are several options here. Although it is possible to require direct measurements of gas release, this is an expensive approach and is likely to discourage projects from being developed. Results from direct measures will also vary based on type of instrument used to collect gas samples, frequency of collection and time of day for collection. Alternatives to direct measures include default factors or project controls.

Reported emissions of CH_4 and N_2O from composts are shown in Table 9 (Brown et al., 2007). In general N_2O emissions were below 1% of total N in systems that included active management (turning or aeration). Methane emissions were reported but it is important to note that the studies with significant CH_4 emissions summed total gas concentrations throughout the pile depth and did not account for potential CH_4 oxidation (Hao et al., 2004, Hao et al., 2001). For studies that measured CH_4 from the surface of the composting pile, minimal quantities of CH_4 were detected (Sommer and Moller, 2000; Kuroda et al. 1996; Lopez-Real and Bapatista, 1996). A recent report compared windrow and in vessel systems and found that in vessel systems released 30% less CH_4 and 38% less N_2O as windrows (Cuhls et al., 2008).

Recent studies have also focused on process controls to minimize emissions. These include covering compost piles with finished compost to oxidize CH_4, incorporating finished compost into active piles to minimize production of NO_2 and so also inhibit formation of N_2O, and keeping piles at 55° C to reduce ammonia formation which will also inhibit formation of N_2O. Recent studies have also confirmed release of N_2O from landfills. However, to date there isn't sufficient information to quantify these emissions based on type of feedstock.

If default factors are used for fugitive emissions from composting a value of 0.5-1% of total N in the finished compost would likely be appropriate. By using total N as a basis, the default value will take into account the reduced potential for N_2O release from composts with lower nitrogen content. This value is potentially lower than the default value used in the Alberta protocol. A lower value is justified based on the recognition that landfills are a source of N_2O. As this is not included in the accounting, a lower baseline for N_2O emissions from composting is justified. Default values for CH_4 emissions are also possible. Here, it is likely that the credits associated with projects will be sufficiently conservative as to underestimate CH_4 release from landfills. As CH_4 is produced exclusively from anaerobic decomposition and as composting is an aerobic process, fugitive emissions of CH_4 from composting operations are likely to be minimal in comparison to landfill emissions. If, despite these factors a decision is made to use default emissions, an appropriate factor for CH_4 would be 2 kg CH_4 per Mg finished compost. Both of these default emissions factors could be used for windrow systems with reduced emissions (based on the Cuhls et al., 2008) study given to in vessel systems.

Process control requirements offer an effective and potentially better alternative to default factors for emissions during composting. Recent research has identified cost effective and relatively straightforward means to minimize emissions from composting operations. The CCX protocol opted to use process control measures in lieu of default factors. For that protocol, all qualifying projects are required to meet US EPA time and temperature standards for pathogen reduction. This assures aerobic conditions which limit the potential for CH_4 formation and requires active pile management. A recent study also suggests that the temperature required for pathogen destruction, 55° C, maximized the rate of decomposition while reducing NH3 emissions (Elkind et al., 2007). This is likely to result in reduced N_2O emissions. Other studies have shown that capping piles with finished compost for the initial phase of composting reduces methane emissions. If the finished material is then incorporated into the pile, research suggests that N_2O emissions will be minimized (Fukumoto et al. 2006). By requiring composting operations meet the EPA requirements for pathogen destruction and that windrow systems cover piles with finished compost during the initial week of composting with finished compost subsequently incorporated into the piles, the potential for fugitive emissions can be minimized. These process controls are also not likely to discourage potential projects.

7.2. Compost GHG Potential

7.2.1. Methane Generation

Methane emissions from composting are more likely to occur early in the composting process and for piles with higher moisture contents. The early phase of composting is characterized by rapid decomposition of organic feedstocks. This aerobic decomposition reduces oxygen concentration in the piles and generates CO_2 and H2O. This phase also produces high heat that leads to evaporation. Methane emissions decline as piles dry out and decompose (Brown et al., 2008). In a windrow system for example, the highest concentrations of CH_4 were observed near the bottom of the windrow, with highest release occurring during turning (Hao et al., 2001). Methane emissions from different types of compost systems range from below detection to 2.5% of initial carbon (Brown et al., 2008). The highest value was derived from summing CH_4 concentrations through a compost pile at 14, 40, 70 and 100 cm below the pile surface and is likely not representative of actual CH_4 release into the environment (Hao et al., 2004). Other studies have noted that CH_4 formed within a compost windrow is generally oxidized in the aerobic upper portion of the pile by methanotrophic bacteria that convert CH_4 to CO_2, mitigating the release of CH_4 to the atmosphere. Consequently, compost has been used as landfill bio-cover material because of its ability to effectively oxidize CH_4 (USEPA, 2006). Storage of finished compost has been shown to release only trace quantities of CH_4 and N_2O (Hao, 2007).

7.2.2. Nitrous Oxide Generation

In the initial phases of composting, nitrogen is present primarily in organic forms. Nitrous oxide can only be formed after organic nitrogen has been converted to mineral forms. As a consequence, N_2O emissions are generally only detected after the initial phase of composting. Hao et al. (2004) monitored gas production at different depths of manure compost windrows. Total N released as N_2O was 0.08 kg/Mg of feedstock. Nitrous oxide concentrations increased in the center of the pile during both the middle and later phases of composting. High moisture contents coupled with low C:N ratios appear to increase the potential for N_2O release. In a study with a high moisture content (65%), N_2O release was significantly greater totaling 46.5 g N_2O/kg N or 4.6% of total N (Fukumoto et al., 2003). This was in a static pile with no aeration. Fukumoto et al. (2006) composted pig manure at a C:N ratio of 7:1 in a static pile with 72% H2O and released 9.9% of total N as N_2O. One study where biosolids were composted in an aerated static pile using wood ash as a bulking agent and a compost moisture content of 75% also showed high emissions of N_2O (Czepiel et al., 1996). In this study 1.8% of total N was released as N_2O.

In studies where the moisture content of the pile was optimized to reduce the potential for anaerobic conditions, N_2O release was much less significant. For example, Sommer and Moller (2000) composted pig litter with low straw content (76% moisture) and pig litter with high straw content (35% moisture). In the low straw pile, 0.8% of initial N was released as N_2O. In the high straw pile, N_2O was below detection.

Reported emissions of CH_4 and N_2O from composts are shown in Table 17 below (Brown et al., 2007).

Table 17 – CH_4 and N_2O Emissions from Compost

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%	Below detection	Below detection
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 $\mu L\ L^{-1}$ 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 (1.8% 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

7.2.3. *Compost Methods*

More recent studies have conducted side by side tests of different composting systems to compare emissions directly between composting systems. This offers the opportunity to make direct comparisons between systems and develop appropriate management tools to minimize emissions of fugitive gases during composting.

Cuhls et al. (2008) shows emissions from different compost systems where biowaste, yard waste and biodegradable commercial waste were used as feedstocks. Here also, values are reported on a wet weight basis. As such, these results should be used to compare different composting systems rather than as a basis for absolute emissions values for different feedstocks or composting systems.

- In vessel and combination of in vessel and open composting plants
    a. CH_4 mean 710 g/Mg range 300 to 1,500 g/Mg
    b. N_2O mean 68 g/Mg range 49 to 120 g/Mg
    c. NH3 mean 63 g/Mg range 15 to 110 g/Mg
- Open windrow composting plants
    a. CH_4 mean 1,000 g/Mg range 470 to 2,000 g/Mg
    b. N_2O mean 110 g/Mg range 49 to 210 g/Mg
    c. NH3 mean 470 g/Mg range 230 to 920 g/Mg
- Composting plants with dry anaerobic digestion and aerobic post-treatment
    a. CH_4 mean 3,700 g/Mg range 3,200 to 4,600 g/Mg
    b. N_2O, mean 120 g/Mg range 38 to 190 g/Mg
    c. NH3 mean 200 g/Mg range 25 to 320 g/Mg

These results suggest that windrow systems may have higher rates of CH_4 and N_2O emissions in comparison to in vessel or in vessel + open composting operations.

7.3. Process Control Techniques or Methods

Recent studies have also attempted to reduce emissions of fugitive gases during composting and landfilling by using different management schemes or process controls. It is likely that fugitive emissions from compost can be controlled or minimized by altering management practices.

7.3.1. All Composting Systems

Porosity

Assuring adequate porosity allows an operator to manage the amount of airflow into a composting mass. Compost guidance recommends 30 to 60 percent air volume. This can be equated roughly to bulk density of 750 – 1,100 pounds per cubic yard. This may be difficult to maintain in a passive pile and will depend on a number of factors (residence time, pile height, material moisture content, etc.)

Material Receiving

Compost feedstocks can begin to decompose and release gasses during storage. First in/First out processing This will ensure that materials do not sit for long periods of time before being processed (as necessary) and added to an active compost pile.

Comply with PFRP

The Process to Further Reduce Pathogens was designed to assure that the core of a composting mass achieves high temperatures for a period of three days in aerated static piles. In windrow systems, the requirement is for 3 days at 55 C for 5 turns or a total of 15 days to assure all parts of a pile re exposed to high temperatures. Complying with PFRP will help to assure adequate mixing and temperatures. As CH4 is generally detected only at the beginning of the composting process, this is sufficient to minimize emissions of this gas.

Adequate moisture

Most composting literature is consistent that adequate moisture for composting is between 40 and 60 percent. Above 60 percent the risk of anaerobic conditions is high. If compost piles require irrigation to maintain sufficient moisture for microbial activity during the initial phases of composting, the potential for CH4 release is minimal.

7.3.2. Windrows

Regular turning

Turning assures that all materials in a windrow are exposed to the high heat of the interior of the pile. Turning also reestablishes pile structure, reinvigorates the pile microbiology and breaks up air and water channels. Turning also redistributes moisture in a pile.

Aerated Static Pile

Blowers adequately sized If the blowers are adequately sized, the composting mass should get a proscribed amount of fresh air to ensure high oxygen levels in the pile, minimizing the possibility for methane generation. In negatively aerated systems, the air can be filtered through a biofilter which may oxidize any methane hat is removed by the aeration system. Some membrane style ASPs have shown to be effective at trapping VOCs.

7.3.3. Storage

As the bulk of CH_4 is produced during the initial phase of composting, CH_4 emissions during storage of finished compost would be expected to be minimal. Nitrous oxide is generally found after the initial stages of composting with concentrations diminishing over time (Fukumoto et al., 2007; Hao et al., 2001). These factors suggest that fugitive gas emissions from storing finished compost would be minimal. Research has demonstrated that this is correct (Hao, 2007).

7.3.4. GHG Reduction Potential

Table 18 provides greenhouse gas reduction estimates associated with various process controls from a number of studies.

Table 18 – GHG Reduction Potential from Various Composting Process Control Measures

Fugitive GHG	Composting Process Control	GHG Emissions Reduction Potential	Source

Fugitive GHG	Composting Process Control	GHG Emissions Reduction Potential	Source
CH_4	Keep moisture content of pile <60%	Elimination	Sommer and Moller, 2000
	Cover surface with 15 cm or more of finished compost	60-80% reduction	Jerry Bartlett, Cedar Grove; Fatih Buyuksonmez, San Diego State University
	Meet US EPA time and temperature requirements for pathogen reduction	This temperature is associated with rapid decomposition, and loss of moisture which will result in aerobic conditions.	Eklind et al., 2007
N_2O	Reduce ammonia formation by keeping temperature at about 55 C	50% reduction in ammonia at 55C in comparison to 67 C	Eklind et al., 2007
	Reduce NO2 concentrations by mixing finished compost into pile	73%	Fukumoto et al., 2006
	Have initial mix C:N ratio > 30:1	Studies with low C:N ratio saw higher release of N2O than studies with high C:N ratio	Brown et al., 2008
	Keep moisture content of pile <60%	Elimination	Sommer and Moller, 2000

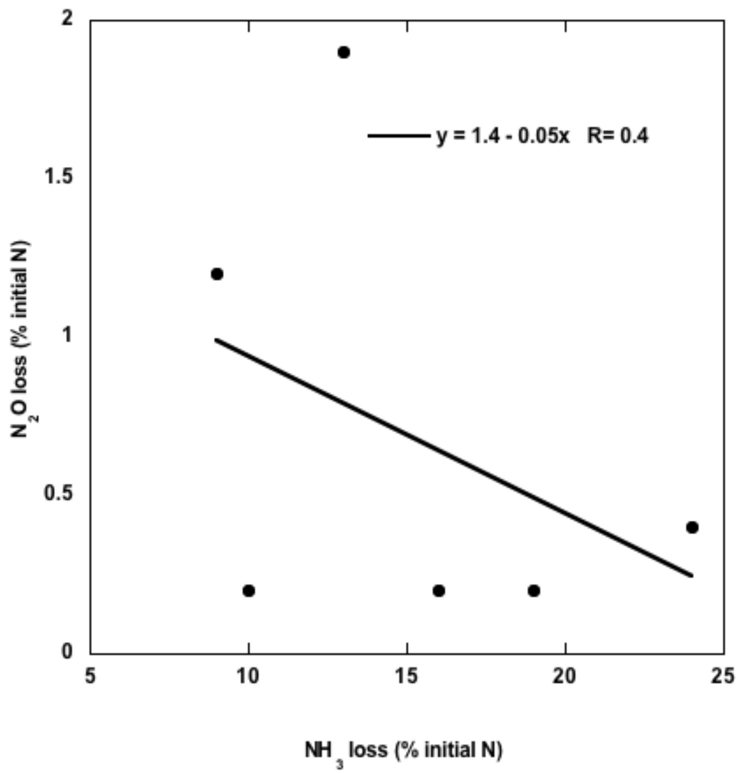
Minimizing N_2O Emissions

Zhang et al. (2009) incubated cover soils collected from 3 landfills in order to establish a relationship between CH_4 concentrations and associated populations of methane oxidizing bacteria and concentration of N_2O. The authors note that the process of oxidizing NH_4 to NO_3- is similar to oxidizing CH_4 to CO_2. Nitrous oxide can be released during the oxidation of ammonia to nitrate. They argue that the microbial community that is responsible for oxidizing methane will also use N as an energy source and inadvertently release N_2O. This suggests that reducing ammonia concentrations is a potential means to limit N_2O emissions from composting operations. A range of options have been proposed to reduce ammonia as well as N_2O emissions.

Eklind et al. (2007) tested ammonia emissions and decomposition rate of composting of source –separated household waste at 40, 55 and 65° C. They found that composting at 55° C maximized the decomposition rate and had half of the ammonia emissions of the 67° C process. Nitrous oxide and methane emissions were not reported. However, another study looked at different composting techniques to reduce N_2O emissions including limited turning, wetting and compacting (El Kader et al., 2007). For this study both cow manure and bedding (low nitrogen) as well as turkey manure and bedding (high nitrogen) were tested as compost feedstocks. Compaction and water addition reduced the rate of decomposition, and N_2O emissions. These practices also likely increased CH_4 emissions; CH_4, however, was not measured in this study.

This study found no relationship between ammonia concentrations and N_2O emissions. In fact, N_2O emissions appeared to decrease with increasing NO_3- concentrations. The results are shown in Figure 15 below. While this study found no relationship between ammonia concentrations and N_2O emissions, N_2O emissions appeared to decrease with increasing NO_3- concentrations, as shown in Figure 15 below.

Figure 15 – Reduction in Nitrous Oxide due to Management Controls



Fukumoto et al. (2006) also focused on the nitrification process in an attempt to control N_2O emissions from pig manure composting. The authors suggest that, in particular, the accumulation of NO2- (the initial microbial product in ammonia oxidation) resulted in release of N_2O. In soil systems, build up of excess nitrite (NO_2-) is recognized as toxic to the bacteria that convert NO_2- to NO_3-. In this study compost piles were inoculated with nitrite oxidizing bacterial or mature pig compost at the end of thermophilic phase to reduce build up of NO_2- and to potentially also reduce emissions of N_2O. The compost piles were maintained in enclosed chambers with air from the chambers continuously monitored for N_2O. Results show that mixing mature compost (with a high concentration of nitrite oxidizing bacteria) was very effective at controlling N_2O emissions. The N_2O emission rates from the three treatments are shown in Table 19 below.

Table 19 – Nitrous Oxide Emissions from Three Treatment Methods

Treatment	**Emission rate g N-N_2O kg^{-1} total nitrogen**
Control	88.5

Mature pig manure compost (MPMC)addition	17.5
Cultured nitrite oxidizing bacteria from MPMC addition	20.2

It appears that mixing compost with finished product during the initial composting phase, maintaining aeration, and having a C:N ratio > 15:1 are effective means to limit release of N_2O. Results from two studies are shown in Table 20 below.

Table 20 – Limiting N_2O by Mixing Compost with Finished Product

Reference	Feedstock	System	% Moisture	C:N Ratio	CH_4 loss	N_2O loss (% of initial N)
Fukumoto et al, 2006	Pig manure + sawdust	Windrow	62	not reported	not reported	
		Control				8.8
		Mixed with 10% aged pig compost				1.75
		Mixed with 1% cultured nitrite oxidizing bacteria				2
Szanto et al., 2007	Pig manure + straw	static pile – no aeration	72	7	12.6 % of VS	9.9
		Windrow		13	0.4 % of VS	2.5

CH_4 Oxidization

Controlling CH_4 emissions can be accomplished by certain management strategies during composting including maintaining sufficient aeration, reducing the moisture content of the compost pile, treating exhaust air with a biofilter and covering the pile with finished compost during the initial stages of composting (Brown et al., 2007). The strategies can be divided into two groups. One emphasis is on preventing formation of CH_4 by maintaining well aerated conditions throughout the composting process. The second is based on assuring that any methane that is formed is microbially oxidized before it is released. These strategies can also be combined.

Well aerated conditions can be maintained in a compost pile by setting up a pile with a sufficiently high % solids and by introducing air into the pile through forced aeration or adequate porosity (Brown et al., 2008). In a text on composting, Haug (1993) discusses moisture contents of different feedstocks and notes that if a pile is too wet it is difficult to maintain an aerobic environment that is required for composting. El Kader et al. (2007) looked at different management practices for composting in an attempt to reduce N_2O emissions. In the well aerated pile 43% of the initial mass was decomposed during composting. In the compacted and wetted pile, only 21% of the initial mass was decomposed. Szanto et al. (2007) composted pig manure and straw with an initial moisture content of 72% in both static pile and windrows. In the static pile 12.6% of the VS degraded was released as CH_4, while in the windrow 0.4% of the VS degraded was released as CH_4. In the static pile 40 ± 5% of the initial organic matter had decomposed while 57 ± 3% of the initial feedstock had decomposed in the well- aerated pile. Hao et al. (2001) shows increased temperatures for extended periods with a turned pile in comparison to a static pile with no aeration. The turned pile also showed much more rapid dissipation of CH_4 and less formation of N_2O.

Covering compost with finished compost or using a biofilter containing compost is an alternate or additional management tool to reduce CH_4 emissions. The use of finished compost as a landfill bio-cover has been shown to oxidize CH_4 and is a recommended GHG reduction practice by US EPA (Chanton et al., 2009; Scheutz et al., 2009; US EPA, 2006). However, not all biofilters or compost bio-covers will perform equally (Clemens and Cuhls, 2003). One study looked at the efficacy of biofilters for scrubbing CH_4 and N_2O from mechanical biological treatment of MSW (Clemens and Cuhls, 2003). The authors found minimal oxidation of CH_4 and suggest actual formation of N_2O through denitrification reactions in the biofilters. The biofilters in this study consisted of woody material and plant roots. It is likely that the biofilters were not keep at an appropriate moisture concentration as denitrification reactions only occur under anaerobic conditions. This may also explain the observed failure of the biofilters to oxidize the CH_4. In other studies, finished compost has been shown to effectively oxidize CH_4 as well as other volatile organic compounds (Abichou et al., 2009; Barlaz et al., 2004; Scheutz et al., 2009; CIWMB, 2007).

From the studies on N_2O and CH_4 emissions from composting, it is possible to recommend several management practices to minimize emissions. Brown et al. (2008) discusses several such practices as shown below.

- 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 (2006b).

- 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.

Recent research suggests alternative means to minimize emissions.

- Capping composting piles with finished compost during the initial stages of composting will eliminate or significantly reduce CH_4 emissions.
- Mixing a biologically active finished compost with composting materials will significantly reduce N_2O emissions
- Letting a pile reach 55° C will ensure aerobic conditions and limit formation of NH3. Reduced NH3 concentrations will limit the potential for NO2- buildup which in turn should reduce potential for N_2O formation during the microbial transformation of NH3 to NO_3-
- In general, maintaining a well aerated pile with a higher C:N ratio will minimize fugitive gas emissions.

The US EPA has standards in place for pathogen reduction in compost systems (www.epa.gov/OWM/mtb/biosolids/503pe/, US EPA, 1993). These rules require that compost systems be maintained at 55° C for specified time periods to ensure pathogen destruction. Based on the literature relating to fugitive GHG emissions from the composting process, the Chicago Climate Exchange requires that all projects that qualify for credits under the methane avoidance protocol, comply with the EPA standards for pathogen reduction. To reach temperatures required for pathogen reduction, piles have to be well aerated. This restricts formation of CH_4 and is also likely to limit formation of N_2O. Covering compost with finished product is also a cost effective means to reduce the potential for CH_4 release.

7.4. Reduction Cost

The economics regarding windrow composting operations that process largely post-consumer food waste depends on several location and industry specific aspects. These include:

1. The dominant economic factor is **Capital and O&M** costs for the operation, including hauling the feedstock to the facility.
2. **Economies of scale** – Above a certain volume (e.g., 50,000 tons/year), windrow composting is less subject to scale than other types of organics processing approaches.
3. **Tipping Fees and Compost Product Value** – Depending on the tipping fees for the materials and the type of product made and the market, the prices may vary and significantly impact the revenue stream of the operation.
4. **Regulatory Controls** – Current and future environmental and health regulations may significantly alter design and O&M costs, such as increasingly stringent controls for pathogens.

Projects at existing solid waste management facilities, but especially at existing composting facilities where they may be incented to modify their operations to obtain carbon credits may have the most favorable economics because project developers only have to invest incremental capital and O&M costs, rather than buying land for a new facility, getting permitted for a new facility and making necessary improvements for eligibility. In some cases, industry owned facilities may be competitive with municipal facilities. For example, large dairies or fruit and vegetable operations that could co-manage other wastes from nearby businesses and/or facilities, could also have the scale to achieve an economic payback. Facilities that combine composting and co-digester operations will also likely have economic advantages because they can use by-products from the digester as feedstock to a compost operation and can manage water and effluent in an integrated way.

Table 21 presents the payback economics for an example composting facility that primarily processes food wastes. The simple payback for this investment of $3 million is approximately 1.7 years. If one considers the value of GHG credits (of avoided methane emissions from MSW being landfilled) estimated at $648,000, the simple payback drops to approximately 1.2 years.

Table 21 – Example Composting Facility Payback Economics

Parameters	Values
Compost Volume	50,000 tons/year
Main Feedstock	MSW -- Primarily Post-Consumer Food Waste
Capital Costs (1)	$3,000,000
Annual Capital Repayment Costs (2)	$180,000
Other Operating Costs (year) (3)	$1,000,000
Total Annual Costs	$1,180,000
Total Annual Tipping Fees (4)	$3,000,000
Total Annual Product Sales	
Total Revenue	$3,000,000
Net Income (before Taxes)	$1,820,000

Source: SAIC

Assumptions:

1) Capital costs no not include design, permitting or real estate costs. Costs based on $60/ton annual throughput.
2) At 6% of capital costs.
3) At 33% of capital costs, includes transportation, 4-6 FTEs, bulking agents, energy, water and environmental costs.
4) Tipping fees are $60/ton.
5) Carbon credit estimated at $10/Metric Ton

8.0 PROJECT BOUNDARY

The GHG assessment boundary delineates the GHG Sources, Sinks, and Reservoirs (SSRs) that must be assessed by project developers in order to determine the net change in emissions associated with a composting project. The definition and assessment of Sources, Sinks, and Reservoirs (SSRs) is consistent with ISO 14064-2 guidance.

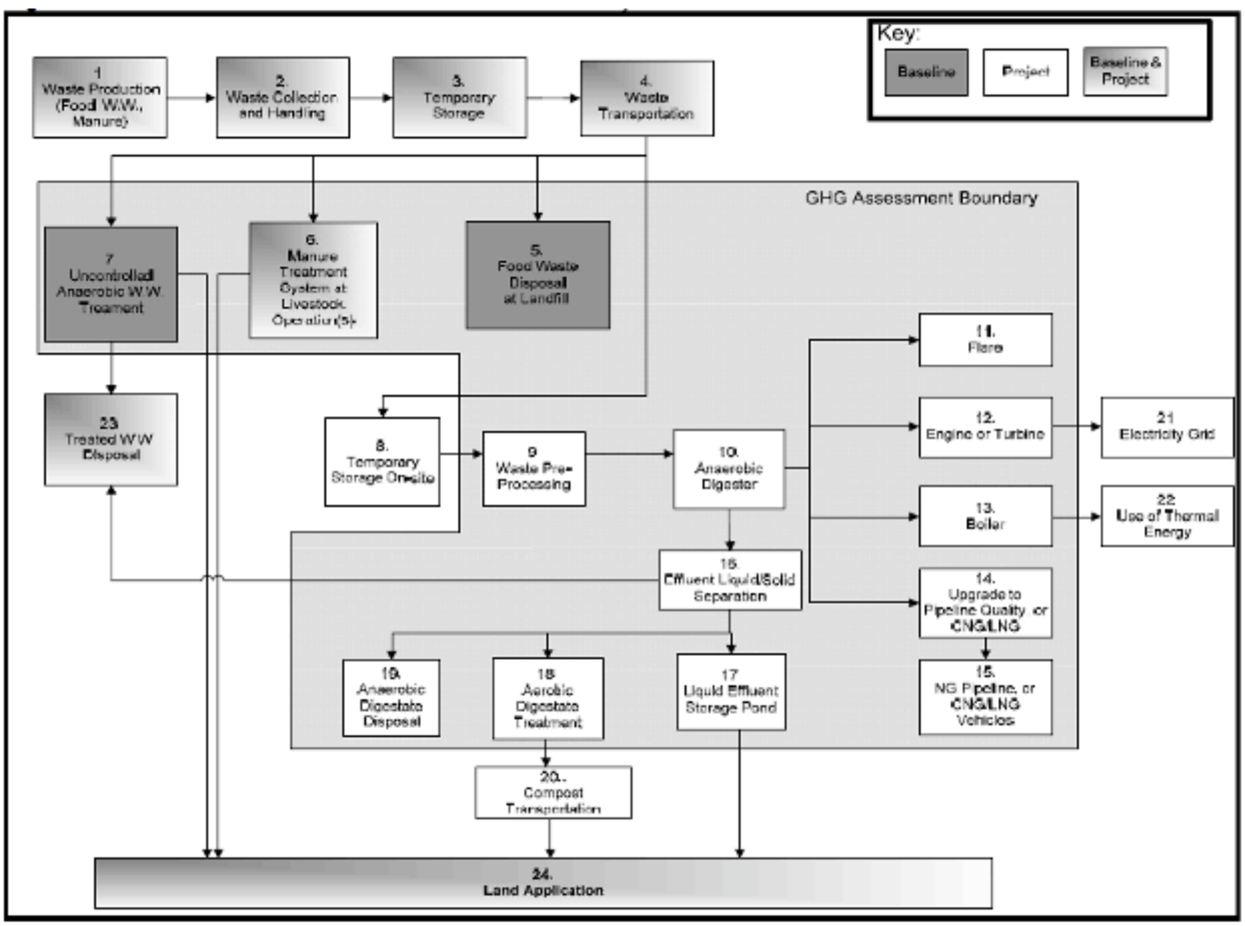
This section discusses how the boundaries for a composting project would be defined in terms of the

- physical boundary of the project
- GHG sources and sinks that should be assessed to determine the net change in emissions attributed to the project activity

8.1. OWD Protocol Project Boundary

The project boundary for the Organic Waste Digestion protocol is illustrated in Figure 16 below.

Figure 16 – Project Boundary as defined in OWD Protocol



8.2. Composting Project Boundary

8.2.1. Baseline Activities

The baseline activities would remain the same as in the OWD Protocol with the generation of organic waste, storage and transportation of that waste being similar to waste that would be sent to a composting facility.

It will be important to note landfill gas collection systems or biogas control systems in place which may capture and destroy the generated methane from baseline conditions as outlined in previous sections of this issue paper.

8.2.2. Project Activities

The activities associated with a composting facility include processing, treatment, handling of the compost and handling of the residue. Other activities include the transportation of the organic waste to and from the facility and the disposal of the finished compost and residue, as well as other electricity and fuel usage to construct, operate and decommission the site and its equipment. In the case of Alberta, treatment and handling of the compost and residues has been included in the project activities. In the case of CCX, treatment and handling of organic waste is included, as is transportation of waste or compost, unless it is determined to be de minimis. At a minimum, the treatment and handling of the compost and residue should be included in the project activities.

The distance required to transport feedstocks to compost facilities is generally similar or shorter than the distance to landfills. Composting operations compete with landfills for particular feedstocks. They could not remain competitive unless transport distances were similar or shorter than hauls to landfills. There are likely to be exceptions to this, however, the economics of composting suggest that shorter haul distances or the potential for a back haul would be necessary to be competitive.

9.0 OWNERSHIP

This section aims to answer the following questions:

- Can ownership of the emission reductions be unambiguously established?
- If not, what are the key issues and how might a protocol address them?

9.1. OWD Protocol Ownership

Indirect emission reductions are reductions in GHG emissions that occur at a location other than where the reduction activity is implemented, and/or at sources not owned or controlled by project participants. The Organic Waste Diversion protocol identifies that projects will result with indirect emission reductions if the project diverts organic waste streams away from landfills or wastewater treatment systems that are not located at the project site or that are not owned or controlled by project participants. Should the project result with indirect emission reductions, it is the responsibility of the project developer to ensure that there will not be claims made (for voluntary or compliance purposes) to the GHG reductions resulting from project activity from the entities that supply organic waste, wastewater or manure to the project. GHG reductions resulting from the digestion project shall be claimed only by the project developer. To ensure that entities supplying waste to the digestion project will not make claim to the GHG reductions resulting from the OWD project, the project developer shall enter into a legally binding agreement with each entity supplying waste to the digester facility. Built into this agreement must be:

1. A commitment by the waste providing entity explicitly granting the GHG rights related to the digestion of the waste stream to the project developer, and
2. A commitment by the waste providing entity to annually complete the appropriate waste source survey (see Section 6.1.4), and
3. A commitment by the waste providing entity to allowing the project developer and the verifier access to the facilities (owned or operated by the waste providing entity) where the waste, wastewater, or manure is generated or treated

9.2. Establishing Ownership of Composting Projects

According to the CCX composting protocol: "By default, project ownership will rest with the facility owner unless it has been awarded by contract to another party". "Projects are defined as facilities that compost organic waste that would have otherwise been landfilled." However, this default assumption rests on the additional assumption that (as is true in most cases) the facility receives `ownership' of the potentially methane-generating resource (the `waste') and therefore assumes the rights to any benefits as well as the burden of any liabilities associated with that resource.

In general, whoever owns the potentially methane producing resource (organic waste) can likely claim rights to any methane mitigation credits (i.e., methane avoidance or reduction) that might occur while the waste is under their control (just like they would own the methane that might be produced in an anaerobic digester, the electricity produced from a biomass to energy project, or, on the down side, the environmental liabilities associated with these resources). If ownership of the resource is transferred, any potential associated benefits or liabilities associated are also transferred (unless explicitly treated differently in a contract).

However, some flexibility and common sense will be necessary in some situations. For example, a generator of waste (household, business, institution, etc.) could conceivably claim ownership rights to credits as long as they didn't transfer ownership (or otherwise dispose) of the material, or if they explicitly retained credit rights through a contractual arrangement with another party. For households and most other generators, once the garbage is hauled away we assume that liability has been transferred as well. This implies that any value associated with the material has been transferred with potential liabilities. The initial 'owner, implicitly or explicitly (depending on the contract) has ceded any rights to further benefits, or risks from liability, related to the resource. However there is a potential that some larger generators (e.g. a supermarket chain) could negotiate a contract with a hauler where they would explicitly retain rights to the credits to be held for use as a revenue source within an investment in composting infrastructure that treats their waste (this type of `stripping' of the environmental assets may make sense if there are other ownership-related permitting or liability issues related to transport and handling of the waste materials). In most cases, however, it would be more straightforward for the generator to simply receive adjusted disposal fees in exchange for the emission reduction credit value of their wastes. Likewise, generators will likely `pay' for the costs of new landfill methane emissions standards indirectly through increased fees.

It has also been argued that landfill owners should be able to claim the rights to any emission reductions related to landfill methane avoidance because emissions `occur at their landfill'. However, this argument is flawed. It is the equivalent of saying that, because traditionally certain waste streams have been landfilled, landfills are de facto owners of these waste streams. However, their ownership of the waste materials is based solely on contractual arrangements with waste generators. If they indeed have contracts that create these kinds of specific obligations with businesses or communities, landfill operators could still only claim damages for non-performance related to the potentially lost value of the mitigation credits or energy resource, not necessarily to the rights to the credits themselves.

10.0 OTHER POSITIVE/NEGATIVE ENVIRONMENTAL IMPACTS

This section briefly discusses the potential for environmental co-benefits from the project activity as well as possible negative consequences.

10.1. Benefits Associated with Compost Use

Compost is applied as a soil conditioner, mulch or to meet the fertilizer requirements for crops. A large number of studies have shown increased soil carbon concentrations when manures, composts or municipal biosolids are land applied (Albaladejo et al., 2008; Favoino and Hogg, 2008; Kong et al., 2005; Pinamonti, 1998; Schroder et al., 2008; Smith et al., 2007). Increasing soil carbon is a cost effective means to sequester carbon that provides a range of ancillary benefits (Lal, 2007). Research has demonstrated increased water holding capacity, increased water infiltration rates, reduced bulk density, improved soil tilth (i.e., health and workability of soil), reduced erosion potential, decreased need for herbicides and pesticides, decreased salinization, reduced fertilizer requirements, and improved yields and/or crop quality (e.g. Cogger et al., 2008; Favoino and Hogg, 2008; Pinamonti, 1998; Recycled Organics Unit, 2006). Each of these can have an enormous financial impact on high value agriculture. In combination, these benefits can result in increased profitability and competitiveness for agriculture.

There are also indications that using compost as a substitute for synthetic fertilizers will reduce N_2O emissions in comparison to synthetic fertilizers (Ball et al., 2004; López-Fernández et al., 2007). A field study conducted on a poorly drained high clay soil in Scotland showed significantly reduced N_2O emissions from compost in comparison to NPK (Ball et al., 2004). Total emissions after 5 applications of amendments (values in kg N ha-1 from the Ball et al., 2004 study) are shown below.

- 26.4 ± 1.29 NPK fertilizer
- 15.3 ± 1.31 cattle slurry
- 10.0 ± 0.67 biosolids compost
- 8.0 ± 1.91 dried pellets
- 10.3 ± 2.12 digested liquid biosolids

The benefits of compost for use in disturbed soils are well documented (Haering et al., 2000; Sopper, 1993). Compost has been used to restore forest lands disturbed by fire (Meyer et al., 2001). Composts have also been used to restore soils impacted by coal mining (Sopper, 1993). Compost use is an integral part of Washington State's best management practices for maintaining water quality and guidelines for improving soils with compost are provided on the Soils For Salmon website (http://www.soilsforsalmon.org/how.htm#bmp). Composts are an integral part of this program as they reduce soil erosion and movement of nutrients to streams. A city ordinance in Leander, TX (Ordinance No. 07-018-00) requires compost to be mixed with topsoil for all new landscapes including residential and non-residential as a way to reduce water usage. Compost use is also recommended by the Bureau of Land Management in Washington and Oregon as a means to restore soils disturbed by mining activities (Norman et al., 1997).

A number of states are now promoting the use of compost as a medium for stormwater management and erosion control. Studies in Texas, Washington, California and others have shown that compost can be an effective medium for reducing erosion on very steep slopes and in places where traditional methods fail. The US EPA has published Best Management Practices for using compost in these applications, including compost blankets, compost filter berms, and socks. These documents can be accessed here http://cfpub.epa.gov/npdes/stormwater/menuofbmps/index.cfm

10.2. Negative Consequences

10.2.1. Odor

Manufacturing compost on a commercial scale typically involves large amounts of organic matter stored and processed outdoors. The predominant technology (windrowing) offers a wide variety of process control from highly managed to less so. Proper siting of facilities is critical to avoid land use conflicts and odor nuisances. In most states composting is a highly regulated activity requiring facility permitting at the local and/or state level. Though in some states yard trimmings composting has been de-regulated somewhat to provide incentives to facility operators and to help implement statewide landfill bans. Composting of food scraps generally has a higher threshold of permitting requirements. California requires that each commercial composting facility prepare and maintain an Odor Impact Minimization Plan. Other states regulate odor with a dilution to threshold standard at the property line. Perception of odors is highly subjective.

There are a variety of management techniques that have been developed to mitigate odor (CIWMB, 2007). Management techniques range from simple to complex. As described above, maintaining adequate airflow is seen as critical to reducing odors. In some cases the process for minimizing potential CH_4 and/or N_2O releases are complimentary to minimizing odors (proper moisture content, adequate oxygen, etc.)

10.2.2. Criteria Air Pollutants

The USEPA regulates "Criteria Pollutants" in air quality and has published thresholds above which impacts to human health may occur. These thresholds are called the National Ambient Air Quality Standards (NAAQS) (US EPA http://www.epa.gov/oar/oaqps/greenbk/o3co.html)

Ozone and particulate matter are two criteria pollutants associated with composting operations. Ozone (the primary component of smog) is formed when Volatile Organic Compounds (VOCs) react with oxides of nitrogen (NOx). Some jurisdictions (including several air districts in California) that are classified as "severe" or "extreme" non-attainment relative to the NAAQS for ozone have developed new regulations pertaining to reducing VOC and/or particulate matter emissions from the composting process itself. VOCs in composting are naturally occurring and a natural part of the composting process, nevertheless in the presence of NOx they may form ozone. Several Air Districts in California have promulgated regulations which pertain to biosolids and manure composting. These feedstocks, when composted can generate significant amounts of ammonia which when combined with NOx can form particulate matter. To date only the San Joaquin Valley Air Pollution Control District (SJVAPCD) has developed a draft rule regulating VOCs from green material composting, though several districts are proposing to develop rules and others have regulated VOCs under existing New Source Review.

The CIWMB has conducted a number of studies on VOCs from green material composting (CIWMB, 2007, 2008). Additional work is being done by the SJVAPCD. Although capture and control (via forced aeration to a biofilter or other control device) is seen as one means to control VOCs, the costs may be prohibitive. Thus the SJVAPCD is currently studying low-cost management practices that may have a similar potential to reduce VOCs. Although the research is pending, the techniques being investigated may also be consistent with reducing CH_4 and N_2O. It is also not clear what the air quality impact of not composting collected organic materials might be. In some cases collected organic materials that could not be composted due to air quality concerns might instead go to landfills. Büyüksönmez (2007) argues that VOCs in uncomposted materials might actually be higher than if they were composted.

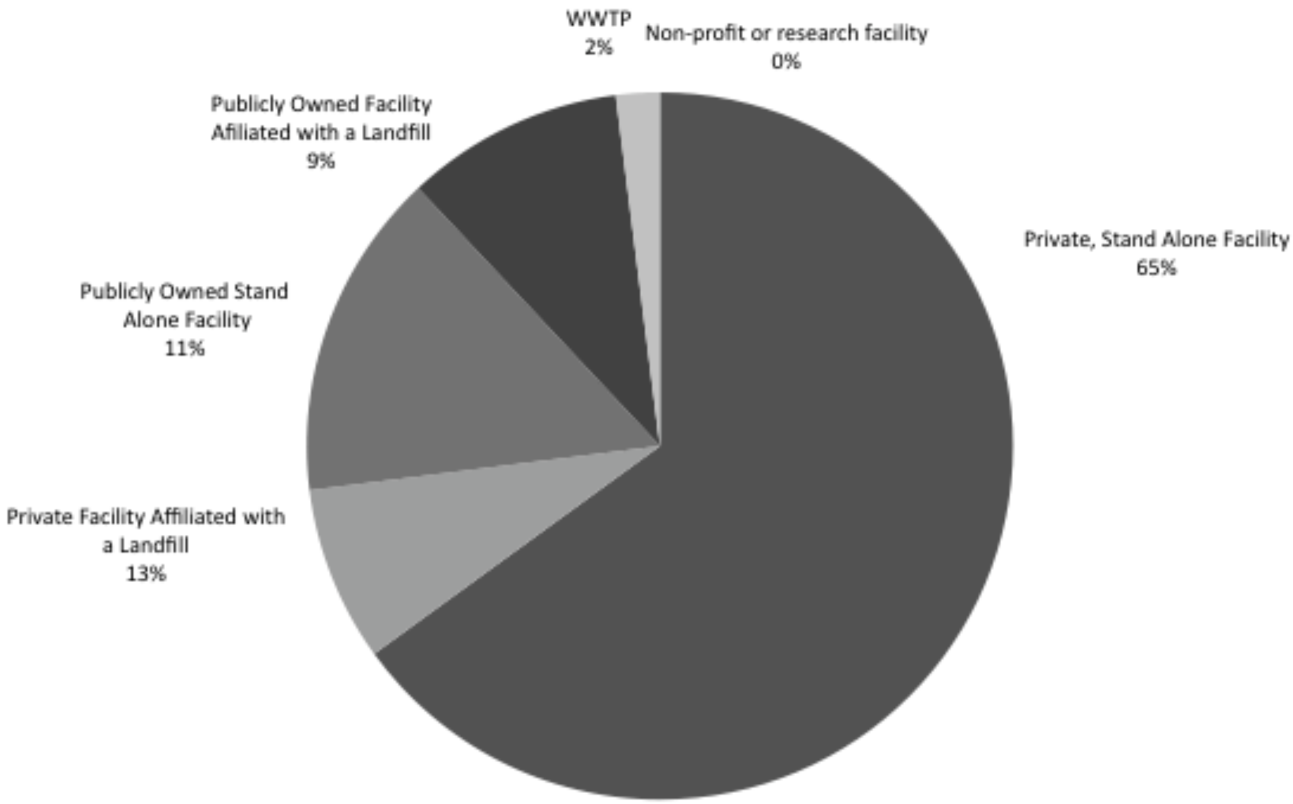
11.0 MARKET INTEREST

This section seeks to identify parties interested in the development of a project protocol for the project activity of diverting organic waste from landfill to a composting facility.

11.1. Compost Facility Ownership

As discussed previously, a recent draft study by CIWMB in 2009 characterized compost facilities in California. Figure 17 shows the breakdown of the types of entities that own composting facilities in CA. The majority of these (65%) are private, stand-alone facilities. The next largest group are privately operated facilities associated with a landfill (13%). .

Figure 17 – Ownership of Compost Facilities



11.2. Potential Compost Project Developers

Compost "project developers" could be categorized into three parties who generate the majority of compost feedstocks:

1. Agribusiness (primarily for manure management)
2. Industrial food processing (manage spoiled/past date food, regular cull and trim and liquid wastes)
3. Municipalities and/or their contractors (waste management companies and private composters) or large institutions (e.g. schools, prisons, hospitals)

For group two much of the waste currently goes to direct land application or cattle feed. The third group (the cities, counties and their contract haulers) are the most likely "interested parties" due to the large proportion of food waste scraps being collected. Statewide diversion goals provide an incentive for project development by municipalities. In addition, the avoided cost of disposal provides a financial incentive. For group two, this is not as strong as an incentive, however regulations for all three of these groups are on the rise (particularly in California) and provide motivation if not incentive to divert waste to compost.

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