



## **Current MSW Industry Position and State-of-the-Practice on LFG Collection Efficiency, Methane Oxidation, and Carbon Sequestration in Landfills**

Prepared For:

**Solid Waste Industry for Climate Solutions  
(SWICS)**

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July, 2007  
File No. 01198086.09 Task 1

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## 1.0 INTRODUCTION AND OBJECTIVES

Methane is a very important contributor to global warming, with its global warming potential 21 to 23 times that of carbon dioxide. Furthermore, methane has a short atmospheric lifetime of about 10 years, so changes in methane sources can affect atmospheric concentrations in a relatively short time scale. Municipal solid waste (MSW) landfills are recognized sources of methane through emissions of landfill gas (LFG).

Current methods used for calculating methane and other pollutant emissions from landfills have been the subject of intense scrutiny. The major concern is that the estimates make assumptions, which do not take into account all relevant factors that can have an influence on actual emissions. In particular, methane oxidation in cover soils, high collection efficiencies for LFG systems, over estimating by United States Environmental Protection Agency (USEPA) models and regulations, and carbon sequestration in landfills are often not accurately covered in the emission inventories and are often debated concepts.

Currently, the solid waste industry is experimenting with various methods to more accurately determine or measure LFG emissions. Due to the high spatial variability of LFG emissions, none of these methods has proven to be practical, and further studies are pending. This white paper identifies the need for updating the LFG emissions protocol in this interim period where statewide inventories are being updated until accurate measurement techniques are developed. The following sections summarize a proposed strategy to update the current inventory protocol.

### 1.1 State of Critical Need

This section of the white paper details the current use of default methods for determining LFG collection efficiency and methane oxidation. By understanding the current status of these concepts, the critical need for update of these methods becomes obvious. With respect to carbon sequestration, this section seeks to identify the critical need for recognition of the ability of landfills to sequester carbon and how landfill greenhouse gas (GHG) inventories should include a sequestration value.

#### 1.1.1 LFG Collection Efficiency

According to the USEPA's "Compilation of Air Pollutant Emission Factors" (AP-42) (USEPA, 1997), researchers and practitioners estimated collection efficiencies to typically range from 60 to 85%. The most commonly assumed default efficiency has been 75% although higher efficiencies have been demonstrated at some sites, particularly those engineered to control emissions.

A review of available data and industry information regarding LFG collection efficiency was conducted by the USEPA in 2002. Eastern Research Group, Inc. (ERG), a USEPA contractor, conducted this review. In an internal memorandum dated October 24, 2002, the USEPA recommended a 75% default LFG collection efficiency (Leatherwood, 2002). Other key points from USEPA's review are included below.

- Overall, minimal data on LFG collection efficiency exist. Industry contacts cited the difficulty in documenting uncontrolled LFG emissions as the primary reason.
- Documenting uncontrolled LFG emissions is problematic because:
  - The high spatial variability of LFG emissions makes it very difficult to accurately quantify LFG emissions.
  - LFG generation rates are highly variable due to the heterogeneity of MSW and variations in rainfall and landfill temperature.
- LFG emission levels are site-specific and vary over time and spatially; therefore, representative collection of samples is difficult.
- LFG systems historically have been satisfied with capturing a majority of the LFG generated to achieve compliance or some other goal. As a result, LFG system owners and/or operators have not been particularly interested in expending additional efforts in trying to determine actual LFG emissions.

Most of the published sources cited by the memorandum are at least 15 years old. Consequently, these sources do not reflect LFG system operational experience after implementation of USEPA's New Source Performance Standards (NSPS; 40 CFR Part 60, Subpart WWW). By December 1998, higher efficiencies were necessary for NSPS Subpart WWW compliance. Most of the collection efficiency estimates in the memorandum were based on speculation.

The only specific claims of calculated collection efficiency included in the USEPA memorandum were estimates attributed to work done by Dr. Stan Zison of Pacific Energy. Applying his patented methodologies to three California landfills with energy projects, Dr. Zison measured collection efficiencies at 85%, 90%, and 95%, respectively. It is expected that gas collection efficiency is higher at NSPS regulated facilities versus sites only designed for energy recovery where applied vacuums are limited to prevent air intrusion and only collect the amount of high quality LFG necessary to meet energy needs.

Furthermore, the default 75% collection efficiency does not take into account the different LFG collection systems that are utilized at landfills. For example, a LFG collection system designed for NSPS compliance is far more capable of higher collection efficiencies than a LFG migration control system. Using a default value of 75% for both of these systems is an assumption that does not have much validity. A default value should take into account the type of collection system employed at the landfill and the regulatory requirements or other drivers for installation and operation.

### **1.1.2 Methane Oxidation in Landfill Cover Soils**

The USEPA AP-42 document categorizes emission sources at landfills as uncontrolled emissions or controlled emissions. Controlled emissions are defined as emissions that are typically controlled by collection of gas through a gas collection system and destruction of the gas through combustion, most typically a flare. As stated in the previous section, uncontrolled emissions from landfills are difficult to estimate. USEPA's AP-42 document uses a theoretical, first-order

methane production model as the basis of the uncontrolled emissions estimate. Equation 5 (shown below) of Section 2.4 of AP-42 shows the formula used to calculate the uncontrolled emissions rate:

$$CM_p = [UM_p * (1-\eta_{col}/100)] + [UM_p * \eta_{col}/100 * (1 - \eta_{cnt}/100)]$$

Where:

$CM_p$  = Controlled mass emissions of pollutant P, kg/yr,

$UM_p$  = Uncontrolled mass emissions of P, kg/yr (from equation 4, section 2.4-4, AP-42 using first-order model)

$\eta_{col}$  = Collection efficiency of the LFG collection system, percent; and

$\eta_{cnt}$  = Control efficiency of the LFG control or utilization device, percent.

The main assumption used in this equation is that all of the gas that is not collected by the LFG control system is emitted into the atmosphere. This equation does not take into account other characteristics of the landfill that do reduce emissions, like methane oxidation into the landfill cover soils. Methane oxidation in the landfill cover soils can dramatically reduce methane. In order to fully address the emissions of a landfill, one needs to take into account the influence of cover soils in the reduction of methane emissions.

Furthermore, the AP-42 states that “average oxidation of methane (on a volumetric basis) in some laboratory and case studies on landfill covers have indicated ranges from 10 percent to over 25 percent with the lower portion of the range being found in clay soils and higher in topsoils.” Due to the uncertainty involved and the lack of a standard method to determine oxidation rate, the USEPA recommends the default factor of 10% by volume methane oxidation for landfills with low permeability cover systems. This is termed a “conservative approach” by the USEPA. This default methane oxidation rate is dated and needs to be updated based on current engineering technologies of landfill cover soils and more recent research on this topic as detailed in this paper.

### **1.1.3 Carbon Sequestration in Landfills**

Carbon storage in landfills can significantly offset GHG emissions from landfills. The decision to include these factors and how they are utilized in a statewide inventory will depend on the accounting protocol employed.

A number of international and domestic protocols including the Intergovernmental Panel on Climate Change (IPCC), the USEPA, the Oregon Climate Trust, and the California Climate Action Registry recognize carbon storage in land filled material as a sink in calculating carbon emissions inventories. These protocols recognize that when wastes of a biogenic origin are deposited in landfills and do not completely decompose, the carbon that remains is effectively removed from the global carbon cycle, or sequestered.

For example, the USEPA has published reports that evaluate carbon flows through landfills to estimate their net GHG emissions. The methodology the USEPA employed recognizes carbon storage in landfills. In these studies of MSW landfilling, the USEPA summed the GHG emissions from methane generation and transportation-related CO<sub>2</sub> emissions, and then

subtracted carbon sequestration (treated as negative emissions). The projected national average of net GHG emissions for landfills was minus 0.02 MTCE/Wet Ton, showing that landfills are “carbon sinks” (USEPA, 1998).

Furthermore, the 2006 GHG emissions published by the California Energy Commission (CEC) indicate that landfill disposal of urban wood waste and yard trimmings is a GHG sink. The report calculated the GHG sink for yard trimming and wood waste, and assumed the only storage would be contained in these waste categories. However, in California, these waste categories represent only 16.4% of the total California waste stream and only 46% of sequestered carbon within landfills; therefore, restricting estimates of carbon storage to only these waste types produces an extremely low value of overall carbon storage for the total amount of waste disposed.

In order to adequately calculate the net emissions from a landfill, all aspects of the carbon cycle as they relate to sequestration and emissions must be addressed. Therefore, the acceleration of carbon storage in all carbon sinks should be a part of any integrated GHG emissions plan to create an accurate GHG emissions inventory for landfills.

## **1.2 Objectives**

Currently, the California Air Resources Board (CARB) is proposing to utilize the following factors in its statewide inventory for landfills:

- 75% default value for collection efficiency where site-specific data are not provided.
- 10% default value for methane oxidation as a percentage of the gas not collected by a collection system
- Carbon sequestration included as an informational item only for the landfill category. Actual reductions for sequestered carbon in wood products are given to the forest products industry regardless whether these products eventually end up in landfills.

The objective of this paper is to provide the current MSW industry position and state-of-the-practice on LFG collection efficiency, methane oxidation, and carbon sequestration in landfills based upon reviewed literature of the industry. Literature was reviewed in order to:

- Evaluate current LFG collection efficiencies at selected landfills and compare them with the AP-42 default factors,
- Evaluate current methane oxidation data in different types of cover soils associated with landfills and compare to default factors,
- Evaluate carbon storage factors (CSFs) calculated for different types of MSW.
- Evaluate the impact of landfill carbon sequestration on GHG emissions accounting and development of reduction strategies and policies.

This paper will also:

- Examine the allowance for more accurate estimates of GHG reductions possible for LFG through higher collection efficiencies and higher methane oxidation capacities through soil covers, and;
- Provide interim techniques for inventorying baseline and annual GHG emissions for landfills until the CEC Study on landfill emissions is completed, and a new emission estimating method is available.



## 2.0 GENERAL BACKGROUND

Under the anaerobic conditions that prevail for most of the life of a typical landfill, approximately one-half of the carbon containing waste is microbiologically converted to a mixture of methane and carbon dioxide gas while the remainder does not appreciably degrade and is considered to be sequestered or stored. The methane and carbon dioxide produced may be collected and flared or converted to energy, which oxidizes the methane to carbon dioxide emitted to the atmosphere. The methane can also be oxidized to carbon dioxide by methanotrophic bacteria in the landfill cover soil. Therefore, the ultimate fate of carbon placed in the landfill is sequestered or emitted as methane or carbon dioxide (Barlaz, 2007). Methane from landfills is considered an anthropogenic source of carbon while the carbon dioxide remains biogenic in origin. Figure 1 shows a not-to-scale schematic of the carbon flow in landfills.

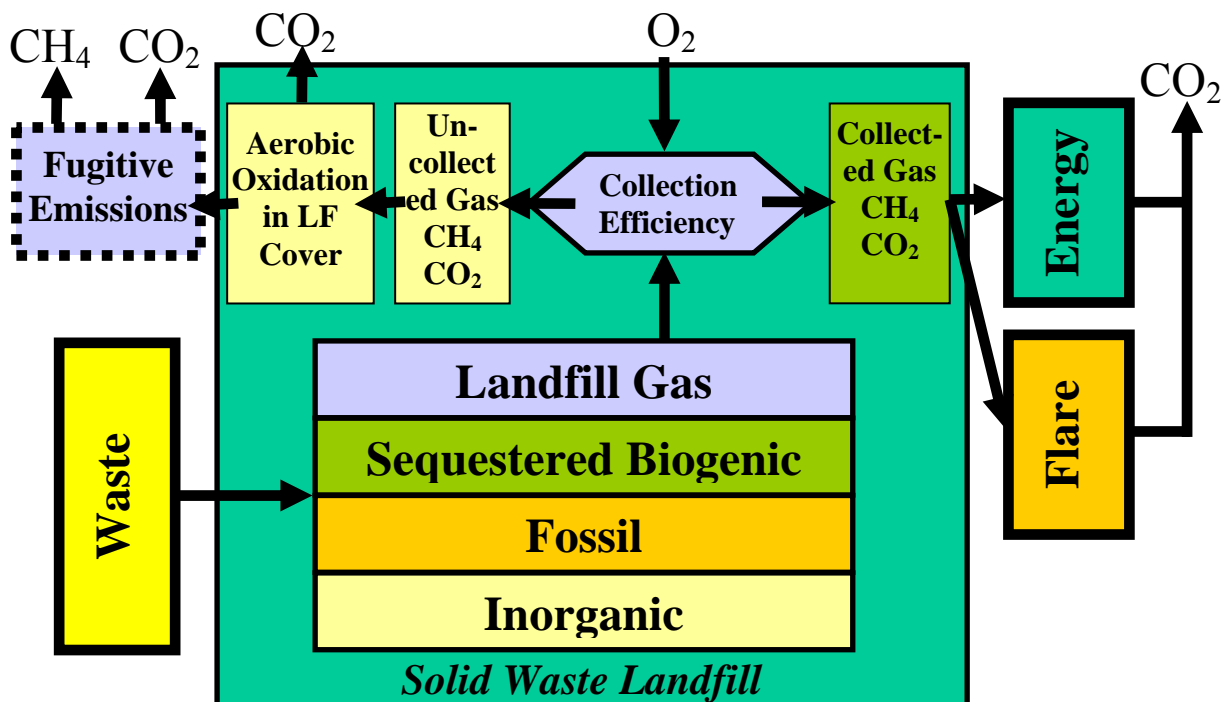


Figure 1. Schematic of Carbon Flow in Landfills

In the United States, as well as in California, MSW is composed of approximately 30 to 50 percent cellulose, 7 to 12 percent hemicellulose, and 15 to 28 percent lignin on a dry weight basis, with cellulose and hemicellulose representing about 90 percent of the biodegradable portion of the MSW (Hilger and Barlaz, 2001). When MSW is buried in a landfill, a complex series of reactions occur in which anaerobic microorganisms decompose a portion of the organic fraction of the waste to carbon dioxide and methane.

Management and treatment of waste ultimately leads to management of the method by which the carbon will be released back into the environment: similarly changing the climate impacts on the way waste will need to be stored, treated, and disposed.

Waste management affects GHG emissions in several ways. The first is landfill methane emissions. The USEPA (1999) suggests that landfill methane accounts for about 4% of all GHG emissions, measured in terms of global warming potential (GWP). The Intergovernmental Panel on Climate Change (IPCC) estimates that landfill methane accounts for 3% of all GHG emissions, but may account for 8 to 10% of all feasible near-term opportunities for emission reduction (IPCC 2001).

The other impacts are less visibly part of the waste management process. Both recycling and waste reduction lead to decreased energy use and process emissions in the industry. Energy recovery from waste displaces fossil fuel consumption. Controlling LFG has two benefits: landfill methane can be substituted for natural gas, a fossil fuel; and combustion converts methane to carbon dioxide, vastly reducing its GHG impact and returning it to a biogenic form.

Paper recycling and reduction has a direct effect on carbon sequestration in forests. Any decrease in the production of virgin paper means that fewer trees need to be cut down, therefore, there may be more carbon left standing in the forests. While a USEPA study finds that there are rather small energy savings due to paper recycling (USEPA, 1998), it also finds that the forest sequestration savings due to recycling or reduction are quite large.

Finally, a noticeable fraction of the carbon in landfilled newspaper, yard waste, and other carbon sources is never released, but remains sequestered indefinitely in the landfill. The inclusion of carbon sequestration in GHG emissions accounting and GHG inventories has been a subject of intense discussion in several published papers since 1990. There is a high degree of uncertainty with respect to methods available for analyzing carbon sequestration in landfills, and there is no universal acceptance regarding whether estimated sequestered carbon should be included as sinks in GHG emissions inventories regardless of the GHG emission accounting method employed (Okereke, 2006). However, as stated above, a number of international and domestic protocols including the IPCC, the USEPA, the Oregon Climate Trust, and the California Climate Action Registry recognize carbon storage in land filled material as a sink in calculating carbon emissions inventories. In light of all the studies and investigations performed, carbon sequestration in landfills should be considered and included in inventories.

In addition, the amount of LFG collected by an active or passive LFG system (i.e. collection efficiency) and the percent oxidation of the remaining methane in the landfill cover soils are also important parameters that must be accurately accounted for in a landfill GHG inventory. This white paper seeks to make recommendations on how each of these concepts should be treated so that the carbon flow in landfills can be estimated to the greatest degree of accuracy possible with current technology.

## 3.0 LFG COLLECTION EFFICIENCY

To recover its energy value and minimize its pollutant emissions, many landfill managers have installed LFG recovery and utilization systems. Recovery of 100 percent of the gas generated is generally considered infeasible due to the permeability of the waste and the inefficiencies, as well as the timing, of installation of the recovery system.

### 3.1 Industry Position on LFG Collection Efficiency

LFG collection efficiency is the amount of LFG (including methane) that is collected in the LFG recovery system, as compared to the amount generated. The USEPA, state, and local regulators use assumed gas collection efficiencies to calculate landfill emissions in regulating and setting policies for landfills. These assumed efficiencies, usually 75%, are much lower than what many gas system operators believe is typically achieved. In many cases, the lower than actual collection efficiencies result from the use of USEPA models that likely over estimate LFG generation, particularly in California where drier climates are commonplace. As previously stated, this reported default collection efficiency is not based on test data and is somewhat dated.

Based upon the literature summarized in Section 3.4, the industry position supports a LFG collection efficiency greater than the 75% default value. A thorough review of various researchers' efficiency determination methods and their results is required to better assess actual collection efficiencies.

### 3.2 Sub-Categorizations for Collection Efficiency Numbers

Numerous publications were reviewed in order to identify the solid waste industries' view on the collection efficiency for LFG. The following sections describe in detail how the engineered controls of the landfill affect the overall collection efficiencies of a LFG extraction system.

#### 3.2.1 Type of Cover

The cover is part of the gas control system (in fact, it may be the only gas control system in old, abandoned landfills with no gas extraction wells), so the potential diffusion or leakage of gas through the cover needs to be quantified. Gas flows through refuse or soils either by advection or by diffusion. Advection occurs when total gas pressure is not uniform throughout the system. Advective flow is in the direction in which total pressure decreases, because gases tend to move from regions of high pressure to regions of low pressure. Diffusive flow of a gas is in the direction in which its concentration (partial pressure) decreases.

Three types of cover are generally used at landfills; final, intermediate, and daily cover. Daily cover is a layer (usually about six inches) that is placed on top of the active fill area of a landfill at the end of each day. Soil is usually used as a daily cover, but other types of materials may be used as well:

- Textile cover (man-made material rolled over the top of the landfill)
- Chemical foams
- Tire chips (tires that have been shredded into small pieces)
- Wood chips or shredded green waste

- Tarps
- Compost

Intermediate cover is used on the top areas of a landfill that will not be used for a long period of time but are not yet ready to be closed. Covering the parts of a landfill that are not used will reduce the amount of liquid that will accumulate in the landfill. Intermediate covers are usually made from any type of soil that is available at the landfill site. The thickness of the intermediate cover is usually one foot, twice as much as is used for the daily cover. Vegetation is usually allowed to grow on top of the cover as protection against erosion.

The final cover on a landfill is meant to seal the landfill and reduce the amount of water that will enter the landfill after it is closed. It usually consists of the below five layers of material, starting at the top of the cover:

- Top soil - This is a layer of nutrient-rich soil. Vegetation is planted here (grass, flowers, etc.). The layer of topsoil is usually about six inches thick.
- Protective layer - The protective layer will protect the barrier layer underneath from cracking, freezing, and thawing during cold months. It will also allow roots from vegetation to grow. This layer is usually made from soil and it is about two feet thick.
- Drainage layer - The drainage layer is used to provide drainage in order to keep as much moisture away from the barrier layer below as possible. This layer is usually made out of coarse sand and it is usually about eight inches thick.
- Barrier layer - The barrier layer is put in place to keep as much water out of the landfill as possible. It is usually made out of clay and it is usually about two feet thick.
- Grading layer - This material is usually very coarse soil or sand. It is put on top of the waste to make the surface uniform and level. This layer is usually six inches to two feet thick.

Along with these three cover types, a synthetic final cover can be used in the form of a plastic liner. The type of cover is directly related to the collection efficiency of the LFG collection system in terms of permeability of the soil or synthetic layer. Thick final clay covers that are compacted obviously have a lower permeability and are more resistant to diffusion of gas (or infusion of air) through the cover than a daily cover soil with a much smaller thickness. Furthermore, plastic liners can basically block all diffusion to the atmosphere, thus resulting in the opportunity for the highest collection efficiencies. This paper will attempt to evaluate and recommend specific collection efficiencies for various cover types and LFG collection systems.

### **3.2.2 Type of LFG Collection Systems**

The effectiveness of a LFG collection system is also dependent upon its design and operation. In general, two types of gas collection systems are in use at landfills; active gas collection systems, and passive gas collection systems. A passive system is a system that relies on pressure or concentration gradients to function (i.e. vertical vents, gravel trenches, etc.). An active system is a system that includes a prime mover that creates a vacuum on a landfill (i.e. vertical gas wells, horizontal collectors, etc.). For the purpose of this paper, only active systems are discussed in further detail. Passive systems, many of which are vented without oxidation of the methane, will be subject to methane oxidation rather than assuming no collection efficiency.

The type of active LFG collection system is directly related to the gas collection efficiency in a landfill. For example, a LFG collection system designed for NSPS (or similar air quality) compliance is far more capable of greater collection efficiencies than a LFG migration control system. Furthermore, a site with a collection system that is used solely for energy recovery is usually not capable of achieving as high a collection efficiency as compared to one that is compliant with NSPS regulations. In summary, the scale or manner of operation of the collection system has a direct effect upon the determination of collection efficiencies.

### **3.3 Proposed New Values for Collection Efficiency**

Based upon the findings in the previous sections and review of recent publications regarding this issue, it is of utmost importance that new values are calculated for LFG collection efficiencies in order to adequately determine emissions from landfills with different types of cover and LFG collection systems. These new values are presented in the bullets below:

- 50-85% (mid-range default = 68%) for a landfill or portions of a landfill that are under daily cover with an active LFG collection system installed but does not have a Resource Conservation and Recovery Act (RCRA) Subtitle D equivalent liner;
- 85-99% (mid-range default = 92%) for a landfill or portions of a landfill that contain intermediate or an engineered final soil cover with an active LFG collection system but does not have a RCRA Subtitle D equivalent liner, and;
- 95-99% (mid-range default = 97%) for landfills that have a RCRA Subtitle D equivalent liner with an active LFG collection system. For the most part, these are modern state-of-the-art landfills that have been designed and constructed from the ground up with modern RCRA Subtitle D equivalent liner systems and gas collection systems that were specifically designed and installed as early as possible in the landfill units operating life. These landfill units were typically constructed after 1991.

The high ends of the range of these values (i.e., 85%, 99%, and 99%) are proposed for sites with NSPS or similar quality LFG collection systems which are designed for and achieve compliance with air quality regulations. The low end would be for full LFG systems that are installed and operated for other purposes, such as energy recovery, migration control, or odor management, or those landfills with surface emission monitoring levels at times greater than 500 parts per million by volume (ppm) per the NSPS; or systems that were retroactively installed well after the landfills operating life had begun. A mid-range default value is provided for those landfills for which detailed information about the construction of the unit is not known.

These values do not address LFG migration control systems that do not provide complete coverage of the landfill (e.g., perimeter system) since the collection efficiencies of such systems can vary over a wide range. For those, an average value of 50% may be appropriate to capture the possible range of efficiencies unless surface emission monitoring levels are less than 500 ppm, then the 85% value is probably appropriate. Also, biologically active materials used as alternative daily cover (ADC) will be assumed to have similar efficiencies as soil covers. This assumption is based upon work performed by Barlaz et al. (2004) and other research summarized in Section 4.0, where biocovers were compared to soil covers in their ability to oxidize methane

as it passes through the covers. It was shown that biocovers oxidize methane just as well as, and sometimes better, than soil covers. Collection efficiency for other ADC types have not been studied are suggested to be subject to a 50% default as well.

### **3.4 Summary of Literature Supporting These Values**

After a thorough review of the literature, publications that addressed the issue of LFG collection system efficiencies directly and provided estimates of LFG recovery efficiencies have been summarized. These publications are discussed below.

EMCON Inc. (EMCON) presented a paper titled “Predicting LFG Generation and Extraction Using the EMCON Model” (EMCON, 1997). This paper provides background information and limitations of the EMCON model for LFG growth and decay cycles. Regarding LFG collection efficiency, the paper states that:

“The extraction system efficiency is a constantly changing value as new refuse is placed daily and a new phase of an extraction system will be placed infrequently. Only when the final extraction system is installed will the efficiency remain constant. At various points in time during the landfill operations, the LFG extraction system efficiency may be determined by judgment, which considers the combined effect of the extraction system and the cover system. The EMCON model can readily predict the LFG rate of generation and yield, but the LFG extraction system efficiency must be determined by judgment. During landfilling operations, the extraction system efficiency should be in the range of 50 to 85 percent. When the landfill is completed and the final cover is in place, it should be in the range of 80 to 95 percent (the lower value is for a landfill where the final cover does not contain a geomembrane component and has a relatively permeable soil cover: the upper value is reflective of a final cover system with a geomembrane component).”

Spokas, et al. (2006) summarized intensive field studies of the methane mass balance for nine individual landfill cells at three French landfills with well-defined waste inputs. The collection efficiency was calculated as the ratio of recovered gas to empirically modeled gas generation. Efficiencies between 88 and 98% were calculated for sites with completed clay covers similar to those widely used in North America. The study reported direct measurements of collection and emissions, the sum of which, in the absence of any storage changes, is the generation. Recalculating collection efficiency by substituting the sum of collection and emission for modeled generation indicates that the final clay covers performed uniformly well [Montreuil-sur-Barse – 93%, Lapouyade – 93% (summer) and 99% (winter); and Grand’landes – 100%].

Michels and Hamblin (2006) used a general approach to determine LFG collection efficiency at 24 landfills in Wisconsin with active gas collection and control systems by dividing the LFG collected by the LFG that is generated (as calculated by USEPA’s LANDGEM model using default coefficients). From years 2000 through 2004, the LFG collection efficiency improved by 7.7% (from 77.3% in the year 2000 to 85% in 2004). It is known that the LANDGEM model overestimates LFG generation and, as a result, LFG collection efficiencies in this study are underestimated. This 5-year improvement in LFG collection efficiency resulted in 1.2 billion

cubic feet of more LFG being collected in year 2004, had no improvement in LFG collection efficiency been made at all. This also reflects the continued implementation of the NSPS rule.

Huitric and Kong (2006) show that collection efficiency, normally expressed as the ratio of the LFG collection to generation rates, can be equivalently and exactly expressed in terms of average LFG surface concentrations. They estimated the efficiency of LFG collection systems in part using “integrated surface methane” (ISM) concentration data for fiscal year 2001-2002 obtained from the Palos Verdes Landfill in Los Angeles County, which does not have a RCRA Subtitle D lining system but does have a substantial final cover system. ISM methodology divides the landfill surface into roughly one acre grids and each grid is monitored in a loop fashion over a half mile route with readings taken every four seconds using an automated methane analyzer and data logger. In Huitric and Kong’s method, concurrently gathered meteorology data are used with actual LFG collection rate to model the reduction in surface concentrations owing to LFG collection. This modeling employs the USEPA’s Industrial Source Complex (ISC) model, which simulates air dispersion mechanisms to study air quality impacts. The collection efficiency was calculated by dividing the modeled ISM reduction due to collection by the calculated total ISM in the absence of LFG collection. The latter is simply the sum of the modeled ISM reduction due to collection and the measured ISM remaining after collection. As each complete landfill ISM survey required several days to complete and was repeated over four quarters, any storage effects within the landfill were assumed to be negligible. Also, hundreds of boundary probes verified that no emissions were lost laterally and all were accounted for within the landfill surface ISM survey. In Huitric and Kong’s method, any emission reductions due to methane oxidation effects are included in the ISM measurement and so an additional assumption with respect to the amount of methane oxidation is unnecessary. Based on coupling of surface emissions data measurements with the ISC model, the study found that the efficiency of the LFG collection system (including methane oxidation) at the Palos Verdes Landfill is 94 to 96%. The authors concluded that “The commonly assumed default collection efficiency value of 75% is dated and does not reflect modern conditions for NSPS-regulated landfills and other landfills operated for emission control purposes.”

Huitric, et al. (2007) attempted to validate the ISM methodology in a more recent study at the Palos Verdes Landfill by using conventional static flux chamber emission measurements. ISM prescreening was conducted in Spring 2006 to identify ten flux chamber measurement locations chosen to represent both peak and typical emission areas. These spring 2006 ISM prescreening data were also used to validate Huitric and Kong’s methodology, and resulted in a +99% collection efficiency. The flux chambers were monitored for methane at five-minute intervals over 35 minutes. Surface flux chamber measurements found no detectable flux in either the five high emissions (individual locations with highest ISM results) or the five spatially representative areas (areas with average ISM results) at Palos Verdes Landfill. Collection efficiencies based on the flux chamber measurements were calculated to be essentially 100%. This result validates Huitric and Kong’s methodology. The improved collection efficiency (+99% in 2006 vs. 94% to 96% in 2001/2002) is due to the gas system improvements in the intervening years.

In a memorandum to the USEPA that resulted from a review of available data and industry information regarding LFG collection efficiency, Leatherwood (2002) noted that Dr. Stan Zison of Pacific Energy reported the only viable collection efficiency data available. He measured collection efficiency at three landfills operated for energy recovery purposes at 85, 90, and 95%.

It is reasonable to expect that gas collection efficiency is yet higher at NSPS regulated facilities. This is substantially higher than the default 75% that Leatherwood supported in the memorandum.

At sites with engineered gas recovery resulting in low methane fluxes to the atmosphere, field measurements have demonstrated that methanotrophs can consume all the methane transported upward to cover soils and, additionally, oxidize atmospheric methane (Bogner et al., 1995a, 1997c, 1999; Borjesson and Svensson, 1997). Recent modeling for landfill settings has indicated that zero or negative emissions are possible where low methane gradients and threshold concentrations in soil gas are present, implying the presence of a pumped gas recovery system (Bogner et al., 2000). These studies imply that the LFG collection systems are removing large amounts of methane from the landfill, resulting in high collection efficiencies, approaching 100% along with a potential to actually remove atmospheric methane.

### **3.5 Differences between 1990 and Today**

In 1990, only a few air agencies in the country were requiring LFG systems. In California, only the South Coast Air Quality Management District (SCAQMD) had a LFG emissions rule in place and fully implemented by 1990 (SCAQMD, 1985). Most systems were being installed for energy recovery, migration control, or odor management purposes. These systems were generally not as efficient as the NSPS-quality systems of today. Further, there have been improvements in every facet of LFG management from 1990 to today, including system design, equipment, construction techniques, and operations and maintenance. While difficult to quantify, this fact must be considered in completing a baseline 1990 inventory versus inventories for current years.



## 4.0 METHANE OXIDATION

Methane oxidation affects the variability of emissions measured at the landfill surface, and is one of the largest unknown variables in preparing global estimates of landfill methane emissions. LFG may pass through the landfill cover, intermediate cover soils or alternative cover materials (compost) before being released to the environment. This presents a potential that microbes in the soil or cover material (methanotrophs) may oxidize some of the methane in the gas. This oxidation reduces the amount of methane released to the environment. Methanotrophic bacteria are important in controlling emissions of methane from landfill covers (Abichou and Chanton, 2004, Bogner et al. 1997a, Straka et al. 1999, Humer and Lechner 1999, and Dammann et al. 1999).

### 4.1 Industry Position on Methane Oxidation

A report conducted by the USEPA in 2004 stated that “average oxidation of methane (on a volumetric basis) in some laboratory and case studies on landfill covers have indicated ranges from 10 percent to over 25 percent with the lower portion of the range being found in clay soils and higher in topsoils,” (USEPA, 2004a). Due to the uncertainty involved and the lack of a standard method to determine oxidation rate, the USEPA recommends the default factor of 10% by volume methane oxidation for landfills with low permeability cover systems. This is termed a “conservative approach” by the USEPA.

Based upon the literature provided in Section 4.4, the industry position on methane oxidation in cover soils is much greater than the default 10%. The default value of 10% needs to be updated based upon technological advancements in soil engineering and state-of-the-practice applications in cover design as well as recent studies detailed below.

### 4.2 Cover Types Evaluated for Methane Oxidation Rates

As previously noted, there are three main types of covers in use at landfills in the United States: daily cover, intermediate cover, and final cover. Also, some landfills use a biocover to further increase the methane oxidation potential of the engineered cover. Typically, a biocover consists of a highly porous gas distribution layer above the waste, which is overlain, in turn, by a compost-amended layer and an upper vegetative layer. Each of these cover types will be assessed relative to its methane oxidation potential.

### 4.3 Proposed New Values for Methane Oxidation

After review of the literature documenting methane oxidation in cover soils, it appears that methane oxidation in the different types of cover are generally higher than the default 10%. Below is a recommendation of values for methane oxidation capacity of landfill covers given in percent oxidation and oxidation rate per unit of area.

#### Percent Oxidation

- Daily Cover – 22.70%
- Intermediate Cover – 32.08%

- Final Cover – 35.63%
- Biocover – 55.33%

#### Oxidation Rate per Area

- Daily Cover – 45 g/m<sup>2</sup>/day
- Intermediate Cover – 86 g/m<sup>2</sup>/day
- Final Cover – 104 g/m<sup>2</sup>/day
- Biocover – 181 g/m<sup>2</sup>/day

These values were calculated by taking the average values from the various field studies that are summarized in the literature detailed below.

The MSW industry recommends that the oxidation rate per area values be used for emission inventory purposes since they represent a more accurate means of measurement. These can calculate methane oxidation by combining the factors with cover areas by type. This represents an improvement over the use of percent oxidation values, which need methane recovery data and collection efficiency estimates or modeled methane generation rate information to complete a calculation. However, if CARB chooses to continue its use of percent oxidation factors, the values detailed herein are recommended.

#### **4.4 Summary of Literature Supporting These Values**

Many publications were reviewed in order to establish the proposed methane oxidation values that are included in Section 4.3. Of all the literature reviewed, these selected publications quantified methane oxidation in cover soils and biocover in terms of percent and mass per area oxidized. Below is a summary of the literature that became a basis for these proposed oxidation values.

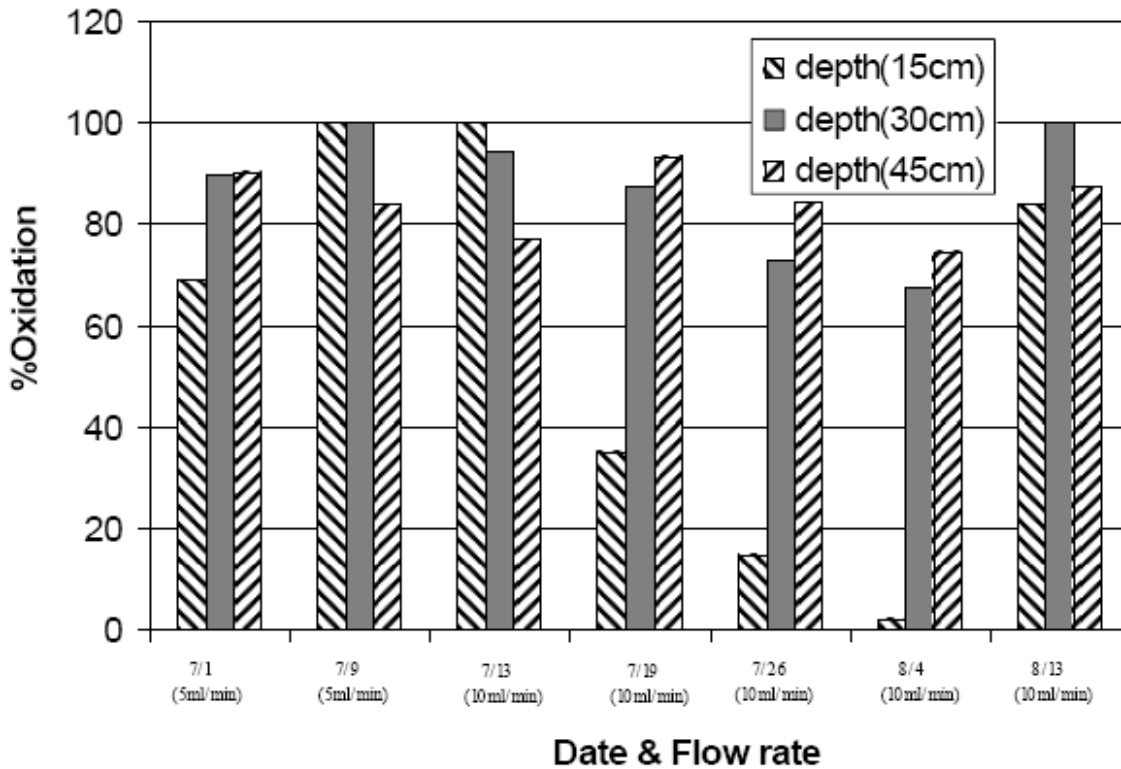
Abichou and Chanton (2004) measured methane emissions at several locations at the Leon County Solid Waste Facility in Tallahassee, Florida, using a static chamber technique. The study area has a composite liner and no gas collection system. The cover soil is sandy clay, varying in depth from about 15 cm to 100 cm, with an overlay of about 15 cm of sandy loam. Three sites with different covers were studied at the landfill: Site 1 consisted of 45-cm-thick (on average) intermediate soil layer (placed on top of 7-year-old waste) with no vegetation. Site 2 consisted of a 45-cm vegetated soil cover placed on top of a 14-year-old waste mass. Site 3 consisted of newly placed waste covered by only non-vegetated daily cover, approximately 15 to 30 cm thick. Methane oxidation at the surface of the landfill varied from 0% to 94.8%, with a mean of 28.6%. The mean degree of oxidation for each site was 22.5% for Site 1, 22.7% for Site 3, and only 11.4% for Site 2 [Because of the relatively small data set for Site 2, (only 2 measurements at 55.7% and 0% were taken), this data point has not been used in calculating the proposed new values for percent methane oxidation]. All of these results show that methane oxidation, even within the daily cover, is above the default value of 10%.

Percent oxidation was estimated after the compost was placed using stable isotope tracing, which has been employed to quantify the oxidation of methane in cover soils (Bergamaschi et al., 1998, Liptay et al. 1998, Chanton and Liptay, 2000, Borjesson et al, 2001, and Christopherson et al. 2001). The below table shows the results of the experiment.

Cell	Average Flux (g/m <sup>2</sup> /day)	Average % Oxidation
No Compost Placed		
2B	14	19
4B	5	17
8B	4	20
Compost Placed		
2D	71	23
4D	3	20
6D	1	40

The results show that oxidation within the squares with no compost is well above the default value of 10%, and with the placement of compost on top of the cover, methane oxidation can dramatically increase.

As part of this study, biofilters to reduce emissions from gas vents in landfills were investigated. Three columns were built using compost with thickness of 15 cm, 30 cm, and 45 cm, respectively. Pure methane (100%) was fed through the bottom of the columns, simulating the methane influx from the solid waste in the landfill through the cover soil. The top of the column was open to allow air diffusion into the column. Columns were designed as 57 cm inner diameter plastic barrels. Gas ports were installed vertically from the surface of the compost at certain depths. Percent oxidation was measured from mass balance calculations for all compost columns. Below is a figure that shows the results of the study:



As shown in the above figure, compost has the greatest oxidation capacity, in most cases, at depths greater than 15 cm, and oxidizes methane at greater rates than the default 10%, given that the compost has adequate gas-filled porosity with high concentrations of methane.

Barlaz et al. (2004) conducted experiments to contrast emissions of methane and non-methane organic compounds (NMOCs) from landfill cells covered with soil or biocover. The soil cover was ~1 meter thick clay over the waste, while the biocover consisted of a 0.15 meter thick clay layer over the waste, followed by 0.15 meters of tire chips, which served as a gas distribution layer, and then 1 meter of yard waste compost. Static chamber tests were conducted on the experimental covers when the gas collection system was turned on and when it was off in order to quantify the methane emissions. Stable carbon isotope analysis was also employed to measure the fraction of methane oxidized as it passes through a landfill cover soil.

Methane flux results show that the biocover consumed atmospheric methane or had near zero emissions. When the gas system was on, uptake of atmospheric methane was measured in 37 of 71 tests and a zero flux was measured in additional 20 tests for the biocover chambers, neglecting April and June 2002 flat section biocover data, which appear anomalous. When the gas collection system was off, 22 of the 36 measurements were negative and 5 of 36 were zero.

Emissions from the soil cover were more variable, with 18 relatively high fluxes (> 15 g/m<sup>2</sup>/day). However, there were also negative fluxes measured in the soil cover even when the gas system was turned off. Of the 50 measurements made on flat and sloped sections of the soil

cover with the gas collection system turned on, 27 were negative and 6 were zero. When the gas collection system was off, only 3 of 25 measurements were negative and 8 were zero.

Calculations of percent oxidation using the stable isotope analysis method ranged from slightly negative (-1.5%) to 103% (full scale). The average percent oxidation for the biocover area (54%) was significantly greater than the average soil cover percent oxidation (21%).

Chanton et al. (2002) also used a stable isotope approach for the determination of methane oxidation in landfill cover soils. The study was conducted by the use of chamber samples from a finished landfill cover (0.5 meters thick topsoil over 1 meter compacted clay separated with a geotextile membrane with a gas extraction system) and a temporary covered area (30 cm sandy clay with a gas extraction system). Isotopic  $\delta^{13}\text{C}$  values of methane were captured in chambers after it had passed across the methane oxidizing bacteria in the cover soil. With the knowledge of the isotopic fractionation factor,  $\alpha$ , they calculated the methane flux and percent oxidation through the cover soils.

The flux of methane across the surface of the landfill within different zones was  $1.97 \pm 0.88$   $\text{g/m}^2/\text{day}$  and  $37.9 \pm 14.6$   $\text{g/m}^2/\text{day}$  for the finished cell and the temporary covered area respectively. The results also show that methane oxidation in the temporary covered area range from 0.2 to 6.1% and 0 to 96% in the finished cell. Average values for percent oxidation were  $3.8 \pm 1.3\%$  and  $40 \pm 7\%$  for the temporary cell and the finished cell, respectively.

Schuetz et al. (2003) examined the emissions of methane and non-methane compounds at a French landfill for two different cover types. One cover was a final cover with vegetation and consisted of 40 cm coarse sand plus 80 cm of loam. The second cover was temporary and consisted of 40 cm of coarse sand. The permanent cover had methane fluxes ranging from -0.01 to 0.008  $\text{g/m}^2/\text{day}$ . The temporary cover had a methane flux of 49.9  $\text{g/m}^2/\text{day}$ . Soil gas profiles indicated that atmospheric air was drawn into the soil cover (Mainly negative flux rates were attributed to the high effectiveness of the gas collection system).

Schuetz and Kjeldsen (2003) investigated the potential of natural oxidation of methane and halogenated organic compounds in soil exposed to LFG in laboratory experiments including both batch and column studies.

For the batch study, soil samples were collected from a landfill in Western Sealand, Denmark, sieved, and placed in a batch container equipped with rubber stoppers, which enable gas to be sampled or injected by a syringe. To obtain methanogenic conditions, air was withdrawn from each container using a syringe and replaced with methane, which gave initial mixture of methane (15% v/v), oxygen (30% v/v) and nitrogen (55% v/v). Gas samples withdrawn from headspace were sampled continuously and analyzed by gas chromatography. From the measured gas concentration, the total amount of test compound in the batch was determined by phase distribution calculations using Henry's Law and the octanol/water distribution coefficient. The kinetics of oxidation were examined by plotting the total concentration of the halogenated compound versus time.

The results of the batch experiments show a high capacity for methane oxidation resulting in very high oxidation rates between 37 and 163  $\mu\text{g CH}_4/\text{g dry soil per hour}$ . Sterilized control

experiments showed no decrease in the methane concentration, which indicates that microbial oxidation is the only explanation for the decrease in the active experiments.

Column experiments simulating a landfill top cover soil matrix through which gas was transported were carried out to examine the degradation process in a dynamic system. The oxidation process was examined in a methane and oxygen counter-gradient system. The columns were packed with landfill cover soil (from the same landfill) and continuously fed in opposite ends with methane gas containing trace components and air. The columns were left with gas for five days before a sampling round; thereafter the experiments were run for at least three weeks in order to obtain steady state conditions. Sampling ports were located along the column length at intervals of 5 cm from the inlet at the bottom. The samples were analyzed using a gas chromatograph. The results of the soil column experiments generally showed a high capacity of methane oxidation giving an average methane oxidation rate of 171 g/m<sup>2</sup>/day corresponding to a reduction of 63%. The control column showed no reduction.

Bogner et al. (1997b) measured methane emissions at an Illinois landfill using static chamber methods. Two contrasting field sites at the landfill were established: GVN (Greene Valley New), which consisted of an interim cover area with recently placed refuse and no gas recovery wells, and GVO (Greene Valley Old), which consisted of a final cover area with abundant vegetation over refuse placed prior to 1980 with vertical gas recovery wells in place. Methane fluxes at GVN ranged from 1.47 to 4.52 g/m<sup>2</sup>/day, while methane fluxes at GVO were much lower, ranging from (-) 0.003 to 0.0008 g/m<sup>2</sup>/day. The lower methane fluxes at GVO can be attributable both to physical factors (thicker clay cover) and biochemical factors (functioning soil microbial ecosystem with high capacity for methane oxidation). Methane oxidation rates from soil cores from a final cover area in the same Illinois landfill were examined by Bogner et al. (1997c). The highest observed rate of methane oxidation was 48 g/m<sup>2</sup>/day, with a range from 16 to 48 g/m<sup>2</sup>/day.

Previous studies at another Dupage County site have also documented that negative fluxes are possible (Bogner et al., 1995b; Bogner et al, 1997c). Recent modeling for landfill settings has indicated that zero or negative emissions are possible only where low methane gradients and threshold concentrations in soil gas are present, implying the presence of a pumped gas recovery system [Bogner et al., 2000]. Thus the combination of engineered and natural controls on landfill methane emissions can be extremely effective in reducing emissions.

De Visscher et al. (1999) performed soil column research utilizing agricultural soils and landfill cover soils. The agricultural soils resulted in a methane utilization rate of 10.7 mol methane/m<sup>2</sup>/day (172 g/m<sup>2</sup>/day), while landfill final cover soil oxidized 15 mol methane/m<sup>2</sup>/day (240.63 g/m<sup>2</sup>/day). These investigators suggest that methane oxidation occurs mainly in the top 30 cm of soil. This proves that methane uptake can have a higher effect on methane emissions than the 10-20% estimated by Czepiel et al. (1996) and rather indicates that the estimate of 50% (45 g/m<sup>2</sup>/day; daily cover) by Whalen et al (1990) can be reached if some landfill cover soil management is implemented.

Kightley et al. (1995) measured the capacity of different soils to oxidize methane in experimental microcosms. Two intermediate cover and one final cover soil samples were collected from

landfill sites within Essex, United Kingdom. Sewage sludge from a nearby sewage treatment plant was also measured for its methane oxidation rate. Below are the results of the experiment.

Soil Type	Methane Oxidation Rate (g/m <sup>2</sup> /day)	% Oxidation
Coarse Sand	166 ± 1.6	61
Clay Topsoil	109 ± 4.8	40
Fine Sand	111 ± 4.8	41
Sewage Sludge	116 ± 2.6	43

Visvanathan et al. (1999) investigated the methane oxidizing capacity of landfill cover soils in Thailand through column experiments. An average oxidation rate of 100 g/m<sup>2</sup>/day for a final cover was achieved. Kjeldsen et al. (1997) investigated the degradation of LFG constituents in LFG-affected soils in a Denmark landfill. The emission of methane was studied in the field by static flux chambers. Relatively high oxidation rates (in the range of 150-250 g CH<sub>4</sub>/g soil/day) were obtained. They were much higher than those reported by Whalen et al. (1990), and Jones and Nedwell (1993). These results were in the same range as found by Figueroa (1993).

Scharff et al (2001) proved that a significant reduction of methane emissions can be achieved through forced aeration of the landfill cover. Oxidation capacities for temporary and final covers averaged 3.0 and 2.0 l/m<sup>2</sup>/hr (48 and 32 g/m<sup>2</sup>/day), respectively. They found a reduction of the methane oxidation capacity with increasing depth, which is caused by a decreasing oxygen flow into the top cover and underlying waste. With increasing depth, the oxygen concentration declines, which results in a less developed methane oxidation capacity.

A test series of column reactors was conducted by Bajic and Zeiss (2001) to simulate methane oxidation in landfill cover soils. Three columns were filled with clay and landfill soil (final cover equivalent), soil and sand (intermediate cover equivalent), and soil and compost (biocover). An influx of gas (45% methane and 45% carbon dioxide, with 10% neon as a tracer gas) was distributed into the columns. The gas concentrations were measured by gas chromatography. The results show that the clay and soil column oxidation rate was greater than the soil and sand column oxidation rate at an average of 45 and 17.5 g/m<sup>2</sup>/day, respectively. The soil and compost column performed better than the others, with a rate of 85 g/m<sup>2</sup>/day, corresponding to 80% reduction in methane. The high porosity of compost allows deep penetration of oxygen into the soil which results that the reaction starts as soon as methane enters the column. It was concluded that oxygen supply is limited by the porosity of the soil/compost.

Powelson et al. (2006) constructed compost biofilters consisting of 238-liter barrels containing a 1:1 mixture (by volume) of compost to expanded polystyrene pellets in order to determine the methane oxidation rate. Gas inflow consisting of an approximate 1:1 mixture of methane and carbon dioxide was introduced into the biofilter and methane oxidation was calculated using the static chamber technique and the Pedersen et al. (2001) diffusion model. An average methane oxidation rate of 242 g/m<sup>2</sup>/day was achieved, corresponding to 69% oxidation. This is similar to

the average methane oxidation of a leaf compost biofilter of 360 g/m<sup>2</sup>/day (69%) found by Wilshusen et al. (2004).

Biocover column laboratory experiments were carried out by Humer and Lechner (2001) at an Austrian landfill. Approximately 150 l of methane/m<sup>2</sup>/day (103 g/m<sup>2</sup>/day) were oxidized at temperatures of 10°C, 18°C, and 30°C. All of the influx of methane into the column was oxidized by the biocover, representing 100% oxidation.

These rates show that methane oxidation can be substantially higher than the default 10%.

#### **4.5 Differences Between 1990 and Today**

Landfill cover practices have also evolved since 1990 with a general trend of improvement after the implementation of RCRA Subtitle D in the early 1990s. For this reason, it is expected that methane oxidation potential has improved with improved (as mandated) cover practices for daily, intermediate, and final cover. Also, biocovers were not in substantial use in 1990 but are today. These factors must be taken into consideration when doing a landfill GHG inventory for 1990 versus more recent years.



## 5.0 CARBON SEQUESTRATION

As stated previously, in the United States, MSW is composed of 30 to 50 percent cellulose, 7 to 12 percent hemicellulose, and 15 to 28 percent lignin on a dry weight basis, with cellulose and hemicellulose representing about 90 percent of the biodegradable portion of the MSW (Higler and Barlaz, 2001). Because of the limited degradation of wood products, significant carbon sequestration occurs in landfills.

Increases in atmospheric levels of carbon dioxide are attributed mainly to fossil fuel burning and deforestation. While efforts to reduce emissions of carbon dioxide continue, increases in carbon dioxide emissions can also be offset, to a degree, by accumulation in carbon sinks such as plant biomass and oceans. It is therefore prudent to focus research efforts both on increasing carbon in sinks and reducing carbon emissions (Skog and Nicholson, 1998). Landfills can and do serve in this capacity.

### 5.1 Summary of How Sequestration has been Allowed for Other Industries

The inclusion of carbon sequestration in GHG emissions reduction and accounting efforts is important because carbon sequestration is a mechanism that could reduce the rate of accumulation of carbon dioxide in the atmosphere by increasing the amount stored in wood products in landfills, plants, or soils. Carbon sequestration concepts in GHG accounting and related guidelines in the Kyoto Protocol have been evaluated by numerous researchers and publications. Marland and Schlamadinger (1999) noted that although the Protocol would sometimes allow emissions credits for sequestered carbon, it would always give credits when fossil-fuel carbon dioxide is displaced. Gillespie (1999) added that the limitation by the Protocol of allowable activities to land-based activities, specifically, reforestation and deforestation, combined with the largely unspecified nature of the inclusion of sequestration and uncertainties in the estimating methodology, contributed to the U.S. decision in 2001 to not participate in the Kyoto Treaty.

The revised guidelines for national emissions inventories published in 1997 by the IPCC stipulate the inclusion of carbon sequestration through land use and forestry in national GHG inventories as an offset to gross GHG emissions from other sources (IPCC, 1997). Below is a table that shows the forestry practices that sequester or preserve carbon.

Key Forestry Practices	Typical definition and some examples	Effect on greenhouse gases
Afforestation	Tree planting on lands previously not in forestry (e.g., conversion of marginal cropland to trees).	Increases carbon storage through sequestration.
Reforestation	Tree planting on lands that in the more recent past were in forestry, excluding the planting of trees immediately after harvest (e.g., restoring trees on severely burned lands that will demonstrably not regenerate without intervention).	Increases carbon storage through sequestration.
Forest preservation or avoided deforestation	Protection of forests that are threatened by logging or clearing.	Avoids CO <sub>2</sub> emissions via conservation of existing carbon stocks.

Forest management	Modification to forestry practices that produce wood products to enhance sequestration over time (e.g., lengthening the harvest-regeneration cycle, adopting low-impact logging).	Increases carbon storage by sequestration and may also avoid CO <sub>2</sub> emissions by altering management. May generate some N <sub>2</sub> O emissions due to fertilization practices.
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Instead of being accounted for in landfills, carbon storage that results from forest products and yard trimmings disposed in landfills is accounted for in the Land-Use Change and Forestry chapter, as recommended in the *Revised 1996 IPCC Guidelines* (IPCC, 1997) regarding the tracking of carbon flows.

Practices such as revegetation, forest management, crop management, and grazing land management have also been allowed as sequestration by subsequent United Nations Framework Convention on Climate Change (UNFCCC) conference of parties. However, inclusion of carbon sequestration by landfilling is not currently included in emissions accounting.

## 5.2 Industry Position on Carbon Sequestration of Landfills

According to Barlaz (2007), an accurate GHG inventory for a landfill would be derived from a carbon mass balance for the landfill and would reflect the net carbon emissions from the landfill. This mass balance approach would account for the amount and types of carbon containing wastes placed in the landfill and their fates over time.

Most carbon sequestration estimates throughout the United States only account for wood waste and yard trimmings within a landfill. Restricting carbon sequestration estimates to only these categories produces an extremely low value of overall carbon storage for total waste disposed. Based upon the literature summarized in Section 5.4, the industry focus on estimating carbon sequestration in landfills is geared towards accounting for all types of carbon containing wastes, not just a selected few.

## 5.3 Proposed Numeric Values for Carbon Sequestration for Landfill Disposal

The 2006 inventory of GHG emissions published by the CEC indicates that landfill disposal of urban wood waste and yard trimmings is a GHG sink. The report calculated the GHG sink for yard trimming and wood waste, and assumed the only storage would be contained in these waste categories. The report used a carbon storage factor (CSF) of 0.26 MTCE/short wet ton of woody waste deposited, and calculated that the GHG sink for lumber and yard trimming disposal was 6.42 million MTCO<sub>2</sub>E for 2003. However, in California, these waste categories (yard trimmings and lumber) represent only 16.4% of the total California waste stream; therefore, restricting estimates of carbon storage to only these waste types produces an extremely low value of overall carbon storage for total waste disposed of about 0.043 MTCO<sub>2</sub>E/short wet ton of total MSW disposed. This is only about 20% of the amount of carbon storage attributed to landfill disposal by Barlaz (1998) and by USEPA (2004). Thus, the total amount of carbon storage in California landfills may be up to five times greater than is currently estimated by the CEC inventory.

It is obvious that there are greater opportunities for carbon storage estimates when all carbon containing waste is taken into account. Below are the proposed carbon storage factors (calculated by the USEPA) that should be used to estimate carbon sequestration at landfills (USEPA 2006).

Refuse Component	Grass	Leaves	Branches	Food	Coated Paper	Old Newsprint	Old Corrugated Containers	Office Paper	Magazines/3rd Class Mail	Municipal Solid Waste
USEPA Carbon Storage Factor (MTCE/Wet Short Ton Refuse)	0.08	0.19	0.31	0.02	0.34*	0.36	0.22	0.04	0.22	0.17
Notes: * - From Barlaz (1998)										

#### 5.4 Summary of Literature Supporting these Values

Most of the literature reviewed to support the inclusion of carbon sequestration of landfills in GHG emission inventories is based on work performed by Barlaz (1998) and the USEPA (2006).

Barlaz conducted laboratory-scale degradation studies to quantify carbon sequestration for the major biodegradable components of MSW including grass, leaves, branches, food, coated paper, old newsprint, old corrugated containers, and office paper. A CSF was calculated that represented the mass of carbon stored (not degraded) per initial mass of the component. The CSF for the MSW components ranged from 0.05 to 0.54 kg of carbon sequestered per dry kg of the waste component.

In addition, USEPA has used these same factors, with some slight modification, and included them into their report: “Solid Waste Management and Greenhouse Gases: A Life-Cycle Assessment of Emissions and Sinks.” The differences between Barlaz’s study and the USEPA’s study are discussed below:

1. The CSF for leaves was set at a lower value for the USEPA because Barlaz’s original value appeared to be too high. Barlaz concurs that the USEPA’s 0.3 gram of carbon per dry gram of refuse is a better CSF for leaves (Barlaz, 2007)
2. Magazines/3<sup>rd</sup> class mail would include some coated paper, for which the CSF is 0.34, and some office paper with a much lower CSF. The USEPA apparently used a weighted average based on assumptions about the relative fractions of magazines and office paper (not all magazines are coated, some are more like newsprint). Barlaz concurs with this adjustment for magazines and 3<sup>rd</sup> class mail (Barlaz, 2007)

Below is a table comparing the two studies:

Refuse Component	Barlaz (1998) CSF (kg C/dry kg Refuse)	USEPA CSF (gm C/dry gm Refuse)	USEPA CSF (MTCE/Wet Short Ton Refuse)
Grass	0.32	0.32	0.08
Leaves	0.54	0.3	0.19
Branches	0.38	0.38	0.31
Food	0.08	0.08	0.02
Coated Paper	0.34	N/A	N/A
Old Newsprint	0.42	0.42	0.36
Old Corrugated Containers	0.26	0.26	0.22
Office Paper	0.05	0.05	0.04
Magazines/3rd Class Mail	N/A	0.26	0.22
MSW	0.22	0.22	0.17

These studies, as well as the literature, indicate that significant carbon sequestration occurs in landfills because of the limited degradation of lignin bearing waste products. These findings are important because they are used in the development of national greenhouse gas emission inventories. In addition, carbon-trading protocols require defensible data on which trades and purchases are based.

### **5.5 Suggestions for How to Incorporate and Use These Data in Statewide GHG Inventories**

For purposes of computing the estimated GHG storage and carbon sequestration in landfills, it is proposed that the CSF from the table in Section 5.3 be used. The CSF will be applied to the tons of waste placed into each state's landfills by refuse type for each year of inventory. If a specific refuse type is not known, a weighted average for the CSF for the entire state's waste stream for a given year should be used. The final step is to convert tons of sequestered carbon equivalents to sequestered tons of carbon dioxide equivalents. This is done by using a factor of the relative molecular weight ratio of carbon dioxide to carbon ( $44/12 = 3.67 \text{ MTCO}_2\text{E/MTCE}$ ). Thus to convert one short ton of material disposed to the relative GHG reduction factor, the following formula should be used (Barlaz, 2007):

$$\text{Short Tons of Material} \times \text{CSF} \times (3.67 \text{ MTCO}_2\text{E/MTCE}) = \text{Sequestered Carbon in MTCO}_2\text{E}.$$

For example, the weighted average computed CSF for the entire California solid waste tons disposed in 2003 is computed to be about 0.11 MTCE/wet ton of refuse disposed. For MSW in California, one ton of MSW disposed would yield the following sequestered amount in  $\text{MTCO}_2\text{E}$ :

$$1 \text{ Short Ton of MSW} \times (0.11 \text{ CSF}) \times (3.67 \text{ MTCO}_2\text{E/MTCE}) = 0.40 \text{ MTCO}_2\text{E}.$$

Use of this factor would result in substantially more carbon storage than is currently estimated by the CEC GHG inventory. If applied to the 40.2 million tons of MSW disposed in California in 2003, a carbon storage factor of 0.11 would result in an estimated carbon sequestration of 16.2 MMTCO<sub>2</sub>E. This is 2.5 times higher than calculated by the CEC and much more than the estimated GHG emissions from landfills that were calculated to be 8.31 MMTCO<sub>2</sub>E in 2003 (Barlaz, 2007).

## 6.0 CONCLUSIONS

In conclusion, if the proposed new values for collection system efficiencies, methane oxidation in cover soils, and carbon storage factors for carbon sequestration are used in the state's GHG emissions inventory, GHG emissions from landfills will be significantly lower and the reduction potential will be higher than the current estimates. The differences in the previously used values and the proposed values need to be taken into account in order to more accurately define the state-of-the-practice technology used in the engineered landfills and the type of MSW landfilled.

### 6.1 Proposed Methodology for Use of Proposed Factors

This section describes how these proposed values for collection efficiency, methane oxidation, and carbon sequestration could be used to replace the current CARB values for collection efficiency (75%), methane oxidation (10%), and carbon sequestration (informational only).

Collection Efficiency – For each year:

- Obtain actual LFG recovery data for the LFG collection system;
- Determine the type of LFG collection system (NSPS/air quality compliance or other);
- Determine the percent of landfill surface under each cover type during the year;
- Determine which landfill areas have RCRA Subtitle D liners; and
- Divide the actual recovery data by a calculated average collection efficiency value based on a weighted average of land surface area by cover type.

Methane Oxidation – For calculating the methane oxidation potential:

- Determine the percent of landfill surface area under each cover type during the year;
- Apply the methane oxidation rate factor for each cover type to calculate the additional methane emission reduction.
- If percent oxidation factors are used, calculate an average oxidation percentage based on a weighted average of land surface area by cover type. Apply this factor to the amount of methane not collected by the LFG system.

Carbon Sequestration – for calculating amount of carbon sequestered:

- Determine the weighted average computed CSF for the MSW in the landfill;
- Determine the amount of MSW landfilled for the year (in short tons);
- Multiply the weighted average computed CSF for the MSW in the landfill by the amount of MSW landfilled for the year (in short tons) to calculate the sequestered amount in MTCO<sub>2</sub>E;
- Include as a separate line item in landfill inventory for avoided emissions by carbon sequestration.

## 7.0 REFERENCES

- T. Abichou, J. Chanton, *Characterization of Methane Flux, Oxidation, and Bioreactive Cover Systems at the Leon County Landfill, Annual Report-* Florida Center for Solid and Hazardous Waste Management. August 2003 to August 2004
- Bajic, Z., Zeiss, C. *Methane Oxidation in Alternative Landfill Cover Soils*. Proceedings from the 24th Annual Landfill Gas Symposium, March 19-22, 2001, Dallas, TX.
- Barlaz, M.A., Ham, R.K, and Schaefer, D.M. **1989**. *Mass Balance Analysis of Decomposed Refuse in Laboratory Scale Lysimeters*. J. Environ. Eng. N.Y. 115:1088-1102.
- Barlaz, M.A., R.K. Ham, and D.M. Schaefer. **1990**. *Methane Production from Municipal Refuse: a Review of Enhancement Techniques and Microbial Dynamics*. Crit. Rev. Environ. Control. 19:557-584.
- Barlaz, M.A. **1998**. *Carbon Storage During Biodegradation of Municipal Solid Waste Components in Laboratory-Scale Landfills*. Global Biochem. Cycles 12, 2: 373-380.
- Barlaz, M. A., Green, R., Chanton, J. P., Goldsmith, C. D. and Hater, G. R. **2004**, *Evaluation of a Biologically Active Cover for Mitigation of Landfill Gas Emissions*, Environ. Sci. & Technol., 38, 18: 4891-4899
- Barlaz, M.A., **2007**. *Recommended Protocol for Estimating Greenhouse Reductions Due To Landfill Carbon Storage and Sequestration*. Report submitted to Solid Waste Industry Climate Solutions Group (SWICS), 2007.
- Bergamaschi, P., Lubina, C., Konigstedt, R., Fischer, H., Veltkamp, A. C., and. Zwaagstra, O. **1998**. *Stable Isotopic Signatures ( $d^{13}C$ ,  $dD$ ) of Methane from European Landfill sites*, Journal of Geophysical Research, 103, D7: 8251-8266.
- Bogner, J., Spokas, K., Burton, E., Sweeney, R., Corona, V. **1995a**. *Landfills as Atmospheric Methane Sources and Sinks*. Chemosphere 31, 9: 4119-4130.
- Bogner, J., and Scott, P. **1995b**. *Landfill Methane Emissions: Guidance for Field Measurement - Final Report to International Energy Agency - Expert Working Group on Landfill Gas*.
- Bogner, J.E., Meadows, M., and Czepiel, P. **1997a**. *Fluxes of  $CH_4$  Between Landfills and the Atmosphere: Natural Engineered Control*, Soil Use and Management, 13: 268-277.
- Bogner, J; Spokas, K; Niemann, M; Niemann, L; Baker, J. **1997b**. *Emissions of Non-Methane Organic Compounds at an Illinois (USA) Landfill: Preliminary Field Measurements*, Proceedings Sardinia 1997 Sixth International Landfill Symposium.

- Bogner J., Burton E., Spokas K. **1997c**. *Kinetics of Methane Oxidation in a Landfill Cover Soil: Temporal Variation, Results of a Whole Landfill Oxidation Experiment and Modelling of Net Methane Emissions*, Environmental Science and Technology, 31: 2504-2514.
- Bogner, J., Spokas, K., and Burton, E. **1999**. *Temporal Variations in Greenhouse Gas Emissions at a Midlatitude Landfill*, J. Environ. Qual., 28: 278– 288.
- Bogner, J., Walter, B., and Sass, R. **2000**. *Model Comparison of Methane Oxidation Across a Management Gradient: Wetlands, Rice Production Systems, and Landfills*, Global Biogeochem. Cycles, 14, 4: 1021– 1033.
- Borjesson, G., and Svensson, B. **1997**. *Seasonal and Diurnal Methane Emissions from a Landfill and their Regulation by Methane Oxidation*, Waste Manage. Res., 15: 33– 54.
- Borjesson, G., Danielsson, A., and Svesson, B.H. **2000**. *Methane Fluxes from a Swedish Landfill Determined by Geostatistical Treatment of Static Chamber Measurements*, Environ. Sci. Technol., 34: 4044-4050.
- Borjesson, G., Chanton, J. P., and Svensson, B. H. **2001**. *Methane Oxidation in Two Swedish Landfill Cover Soils Determined with the Use of  $^{13}\text{C}/^{12}\text{C}$  Isotope Ratios*, Journal of Environmental Quality, 30: 369-376.
- Chanton, J. P., and Liptay, K. **2000**. *Seasonal Variation in Methane Oxidation in a Landfill Cover Soil as Determined by an Insitu Stable Isotope Technique*, Global Biogeochemical Cycles, 14: 51-60.
- Chanton, J.; Bogner, J. **2001**. *Technical Report Prepared for Fentre de Recherches pour l'Environnement l'Energie et les Dechets (CReeD)*, Limay, France.
- Chanton, J.P., Fields, D., Bogner, J., Morcet, M., and Scheutz, C. **2002**. *A Stable Isotope Technique for Determining methane Oxidation in Landfill Covers*, Proceedings SWANA 25th annual landfill gas symposium, March 25-28th, Monterey, California, published by SWANA, Silver Spring, MD.
- Christophersen, M., Holst, H., Kjeldsen, P., and Chanton, J. P. **2001**. *Lateral Gas Transport in a Soil Adjacent to an Old Landfill: Factors Governing Emission and Methane Oxidation*, Waste Management & Research, 19: 126-143.
- Czepiel, P.M., Mosher, B., Crill, P.M, and Harriss, R.C. **1996**. *Quantifying the Effect of Oxidation on Landfill Methane Emissions*, J. Geophys. Res., 101:16,721-16,729.
- Dammann, B., Stresse, J., and Stegmann, R. **1999**. *Microbial Oxidation of  $\text{CH}_4$  from Landfills in Bio-Filters*, Proceedings of Seventh International Waste Management and Landfill Symposium, Cagliari, Italy.



De Visscher A., Thomas, D., Boeckx, P., and Van Cleemput, O. **1999**. *Methane Oxidation in Simulated Landfill Cover Soil Environments*, *Env. Sci.* 33: 1854-1859.

EMCON Guideline. *Predicting LFG Generation and Extraction Using the EMCON Model*. September 1997.

Figueroa, R. A. **1993** *Methane Oxidation in Landfill Top Soils*, Proceedings Sardinia '93, 4th International Landfill Symposium, Cagliari, Italy.

Gillespie, A. **1999**. *Sinks, Biodiversity & Forests: The Implications of the Kyoto Protocol upon the Other Primary UNCED Instruments*, Presented to the International Conference on Synergies and Coordination between Multilateral Environmental Agreements, United Nations University, Tokyo, 14-16 July.

Higler, H., and Barlaz, M.A. **2001** Anaerobic Decomposition of Refuse in Landfills and Methane Oxidation in Landfill Cover Soils, Manual of Environmental Microbiology, 2<sup>nd</sup> Ed., Am. Soc. Microbiol., Washington D.C., pp. 696-718.

Huitric, R., and Kong, D. **2006** *Measuring Landfill Gas Collection Efficiencies Using Surface Methane Concentrations*, Solid Waste Association of North America (SWANA) 29<sup>th</sup> Landfill Gas Symposium, St. Petersburg, FL.

Huitric, R., Kong, D., Scales, L., Maguin, S., and Sullivan, P. **2007** *Field Comparison of Landfill Gas Collection Efficiency Measurements*, Solid Waste Association of North America (SWANA) 30<sup>th</sup> Landfill Gas Symposium, Monterey, CA.

Humer, M., and Lechner, P. **1999**. *CH<sub>4</sub> Oxidation in Compost Cover Layers on Landfills*, Proceedings of Seventh International Waste Management and Landfill Symposium, Cagliari, Italy.

Humer, M., and Lechner, P. **2001** *Microorganisms Against the Greenhouse Effect – Suitable Cover Layers for the Elimination of Methane Emissions from Landfills*, Proceedings from the Solid Waste Association of North America's (SWANA), Publication GR-LM-0006, 6th Annual Landfill Symposium, San Diego, California, pp. 305-318

Intergovernmental Panel on Climate Change, **1997**. Greenhouse Gas Inventory Reference Manual: Revised 1996 IPCC Guidelines for National Greenhouse Gas Inventories, Vol. 3, Paris, France, web site [www.ipcc.ch/pub/guide.htm](http://www.ipcc.ch/pub/guide.htm).

Intergovernmental Panel on Climate Change, **2001**. Summary for Policy Makers of the IPCC WG III Third Assessment Report.

Jones, H.A., and Nedwell, D.B. **1993**. *Methane Emission and Methane Oxidation in Landfill Cover Soil*, *FEMS Microbiol Ecol* 102:185–195.

Kightley, D., Nedwell, D. B. and Cooper, M., **1995**. *Capacity for Methane Oxidation in Landfill Cover Soils Measured in Laboratory-Scale Soil Microcosms*, Appl. Environ. Microbiol. 61: 592–601.

Kjeldsen, P.; Dalager, A.; Broholm, K. **1997** *Attenuation of Methane and Non Methane Organic Compounds in Landfill Gas Affected Soils*, Journal of Air and Waste Management Association, 47: 1268-1275.

Leatherwood, C. **2002**. *Review of Available Data and Industry Contacts Regarding Landfill Gas Collection Efficiency*, Draft Memorandum to Brian Guzzone, Meg Victor, U.S. EPA, October 24, 2002.

Liptay, K., Chanton, J. P., Czepiel, P., and Mosher, B. **1998**. *Use of Stable Isotopes to Determine Methane Oxidation in Landfill Cover Soils*, Journal of Geophysical Research, 103: 8243-8250.

Marland, G. and Schlamadinger, B. **1999**. *Carbon Sequestered, Carbon Displaced and the Kyoto Context*. Proc. TAPPI Int. Environ. Conf. 1:283-289.

Michels, M., Hamblin, G. **2006**. *LFG Collection Efficiency is Improving in Wisconsin*, Waste Management, Cornerstone Environmental Group.

Okereke, V. **2006** *Analysis of Carbon Sequestration at the Cedar Hills Regional Landfill*, Presented at SWANA's 21st Annual Pacific Northwest Regional & Canadian Symposium, April 5-7, Richmond, B.C., Canada

Pedersen, A.R., Petersen, S.O., and Vinter, F.P., **2001**. *Stochastic Diffusion Model for Estimating Trace Gas Emissions with Static Chambers*, Soil Science Society of America Journal, 65: 49-58. [The diffusion model computer program can be obtained from <http://www.agrisci.org/afdelinger/forskningsafdelinger/jpm/medarbejdere/sop>]

Powelson, D., Chanton, J., Abichou, T., and Morales, J. **2006**. *Methane Oxidation in Water-Spreading and Compost Biofilters*, Waste Management Research, 24: 528 DOI: 10.1177/0734242X06065704

SCAQMD, **1985**. *Rule 1150.1 Control of Gaseous Emissions from Municipal Solid Waste Landfills*, South Coast Air Quality Management District, adopted April 5, 1985. <http://www.aqmd.gov/rules/reg/reg11/r1150-1.pdf>.

Scharff, H., Oonk, H., Vroon, R., Van Der Sloot, H., Van Zomeren, A., and Hensen, A. **2001**. *Improved Methane Oxidation by Means of Forced Aeration Under a Landfill Cover*. Proceedings Sardinia 2001, Eighth International Waste Management and Landfill Symposium, Cagliari, Italy.

Scheutz, C., and Kjeldsen, P. **2003**. *Methane Oxidation and Degradation of Organic Compounds in Landfill Soil Covers*, Environmental Science and Technology, 37: 5143-5149.

Scheutz, C., Kjeldsen, P., Chanton, J., Blake, D., and Bogner, J. **2003.** *Comparative Oxidation and Net Emissions of CH<sub>4</sub> and Selected Non-CH<sub>4</sub> Organic Compounds in Landfill Cover Soils*, Environmental Science and Technology 37: 5150-5158.

Skog, K. E., and Nicholson, G. A., **1998.** *Carbon Cycling Through Wood Products: The Role of Wood and Paper Products in Carbon Sequestration*, Forest Products Journal 48:75-83.

Spokas, K., Graff, C., Morcet, M., and Aran, C. **2003.** *Implications of the Spatial Variability on Landfill Emission Rates on Geospatial Analyses*,” Waste Management and Landfill Symposium, Cagliari, Italy.

Spokas, K., Bogner, J., Chanton, J.P., Morcet, M., Aran, C., Graff, C., Moreau-Le Golvan, Y., and Hebe, I. **2006.** *Methane Mass Balance at Three Landfill Sites: What is the Efficiency of Capture by Gas Collection Systems?*, Waste Management, 26, 5: 516-525.

Straka, F., Crha, J., Musilova, M., and Kuncarova, M. **1999.** *LFG-Biofilters on Old Landfills*, Proceedings of Seventh International Waste Management and Landfill Symposium, Cagliari, Italy.

US's Climate Change Action Plan, adopted in 1993.

USEPA, **1997.** Compilation of air pollution emission factors, Report Number AP-42, 5<sup>th</sup> Ed. Supplement C, Office of Air Quality Planning and Statistics, U.S. Environmental Protection Agency, Washington, D.C.

USEPA, **1998.** Greenhouse Gas Emissions from Management of Selected Materials in Municipal Solid Waste, U.S. Environmental Protection Agency, Washington, D.C. (EPA 530-R-98-013).

USEPA, **1999.** Inventory of U.S. Greenhouse Gas Emissions and Sinks: 1990-97, U.S. Environmental Protection Agency, Washington, D.C. (EPA 236-R-99-03).

USEPA, **2004.** Direct Emissions from Landfilling Municipal Solid Waste, U.S. Environmental Protection Agency, Washington, D.C.

USEPA, **2006.** Solid Waste Management and Greenhouse Gases: A Life-Cycle Assessment of Emissions and Sinks, 3<sup>rd</sup> Ed., U.S. Environmental Protection Agency, Washington, D.C. September 2006.

Visvanathan, C., Pokhrel, D., Cheimchaisri, W., Hettiaratchi, J., Wu, J., **1999.** *Methanotrophic Activities in Tropical Landfill Cover Soils: Effects of Temperature, Moisture Content and Methane Concentration*, Waste Management Research, 17: 313-323.

Whalen, S., Reeburgh, W., and Sandbeck, K. **1990.** *Rapid Methane oxidation in a Landfill Cover Soil*, Appl. Environ. Microbiol. 56:3405-3411.

Wilshusen, J.H., Hettiaratchi, J.P.A. & Stein, V.B. **2004.** *Long-Term Behavior of Passively Aerated Compost Methanotrophic Biofilter Columns*, Waste Management, 24: 643–653.