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ESTIMATE OF METHANE EMISSIONS FROM U.S. LANDFILLS

by

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16. ABSTRACT The report describes the development of a statistical regression model used for estimating methane (CH_4) emissions, which relates landfill gas (LFG) flow rates to waste-in-place data from 105 landfills with LFG recovery projects. (NOTE: CH_4 flow rates from landfills with LFG recovery systems can be used as surrogates for CH_4 generation and successively for CH_4 emissions.) The model has three linear segments, each of which applies to a distinct landfill size class. Assumptions were required to account for the recovery efficiency of LFG projects and for the probable oxidation of CH_4 in the top soil cover of the landfill. National CH_4 emissions may be estimated by applying the regression model to municipal-waste-in-place data for U.S. landfills collected in 1986 by EPA's Office of Solid Waste (OSW). This value is adjusted for CH_4 emissions from industrial landfills and CH_4 which is currently recovered or flared. For 1986, CH_4 emissions from U.S. landfills were estimated at 11 tg/yr with lower- and upper-bound values of 7 and 15 tg/yr, respectively. For 1992, estimates were between 9 and 18 tg/yr. The report details uncertainties which limit the quality of the above estimates. The report concludes with a discussion of trends which will affect future LFG emissions, as well as LFG utilization.

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Regression Analysis	Stationary Sources	13C
Earth Fills	Municipal Waste	07C
Wastes		14G
Methane		
Emission		
Estimating		
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ABSTRACT

The estimation of U.S. methane (CH_4) emissions from landfilled waste is part of a bigger effort by the U.S. Environmental Protection Agency's Air and Energy Engineering Research Laboratory (EPA/AEERL), Global Emissions & Control Division, to obtain global greenhouse gas emissions data. Methane flow rates from landfills with landfill gas (LFG) recovery systems were used as surrogates for CH_4 generation and successively for CH_4 emissions. AEERL collected data on 112 U.S. LFG recovery projects, 105 of which are included in the "ORD Database."

The development of a regression model relating LFG flow rates to waste in place data from the ORD Database, is described in this document. The model has three linear segments, each of which applies to a distinct landfill size class. Correction factors were used to account for the recovery efficiency of LFG projects and for the probable oxidation of CH_4 in the top soil cover of the landfill.

In 1986, the EPA's Office of Solid Waste (OSW) conducted a survey, in which detailed information on 1,175 U.S municipal landfill facilities was collected in the "OSW-Westat Database." This population was designed to be a stratified random sample so its data can be extrapolated by means of scaling factors, to obtain total waste in place for active U.S. municipal solid waste landfills. This database contains data which make it possible to estimate waste in place by two different methods. The method which is based on the difference between design capacity and remaining capacity of the landfill, appears to be the more appropriate. The total waste in U.S. landfills in 1986 was $4.7*10^{15}$ g ($5.2*10^9$ tons). The yearly disposal rate (1986–1992) was estimated to be 248 tg/yr (273 tons/yr).

Application of the regression model to the waste in place data calculated from the OSW-Westat Database yields national CH_4 emissions from landfills. Methane emissions from industrial landfills were added to this value and CH_4 emissions which are currently recovered or estimated to be flared were subtracted. For 1986 CH_4 emissions from U.S. landfills were estimated to range from 7 to 15 tg/yr (8 to 16 tons/yr) with a mid-point of 11 tg/yr (12 tons/yr). Methane emissions in 1992 were estimated to be between 9 and 18 tg/yr (10 and 20 tons/yr) with a mid-point of 13 tg/yr (15 tons/yr).

The report details uncertainties which limit the quality of the above estimates. The main uncertainty arises from the inability to perform quality assurance on the OSW-Westat Database, as it exists today. The report concludes with a discussion of trends which may affect future LFG emissions, as well as LFG utilization. Upcoming regulations for controlling air emissions from new and existing landfills are expected to significantly reduce LFG emissions.

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ABBREVIATIONS, SYMBOLS, AND DEFINITION OF MSW

AEERL	Air and Energy Engineering Research Laboratory
BMP	Biochemical Methane Potential
CFM	Cubic Feet per Minute
EPA	Environmental Protection Agency
LFG	Landfill Gas
MSW	Municipal Solid Waste
ORD	Office of Research and Development
OSW	Office of Solid Waste
RCRA	Resource Conservation and Recovery Act
U.S.	United States
g	gram
kg	kilogram, 1,000 grams
Mg	megagram, 10^6 grams
tg	teragram, 10^{12} grams
ft^3	cubic feet
l	liter
min	minute
ml	milliliter
m^3	cubic meter
yr	year
CH_4	methane
CO_2	carbon dioxide
$C_6H_{10}O_5$	cellulose
$C_5H_8O_4$	hemicellulose

DEFINITION OF MSW:

Municipal solid waste includes wastes from residential, commercial, and certain industrial sources, and does not include construction and demolition wastes, sludges, power plant ashes, hazardous wastes and industrial process wastes. (Industrial sources producing MSW may be workshops and other small industries which are typically found among commercial or residential sources, where the waste is comparable to commercial or residential waste in composition, as well as in collection and treatment method.)

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INTRODUCTION

BACKGROUND

Methane (CH_4) produced via the anaerobic decomposition of waste buried in landfills and open dumps is a significant contributor to global methane emissions, with estimates ranging from 10 to 70 teragrams [tg or 10^{12} grams (g)] per year. Global anthropogenic sources emit 360 tg/yr (IPCC, 1992) which suggests that landfills account for 3 to 19 percent of the total. Existing emission estimation methodologies for this source tend to assume that optimal conditions for anaerobic decomposition exist within a landfill. However, this is rarely the case as the information in the following section and an article by Rathje (1991) indicate.

To address this concern, AEERL began a research program in 1990, aimed at using field data from LFG recovery sites to develop an empirical model relating LFG flows to waste in place. The research program started with a review of available models and data and identified several theoretical models and laboratory experiments used to estimate CH_4 from individual landfills. However, these methodologies usually rely on site-specific data which are difficult to extrapolate. Some emission estimation methodologies were found to be reasonable, but the estimates were based on assumed values for certain parameters, such as refuse generation rates and waste composition. In order to develop a new and better model, AEERL initiated a second phase in the program to identify key variables that affect CH_4 generation from buried waste and at developing an empirical model based on those variables. (Thorneloe, 1994a.)

Landfills with gas recovery systems, where LFG is collected and measured by personnel on site, offer a unique opportunity for studying CH_4 emissions. LFG recovery rates can be used to estimate CH_4 generation which in turn can be related to CH_4 emissions. However, in order to use this approach, the accuracy of such LFG data needed to be verified. Furthermore, the availability of additional information on the landfills from which LFG data were collected, including the amount and nature of the waste present, needed to be examined.

The first step in developing this second phase program was a field study of six U.S. landfills with LFG recovery systems (Campbell *et al.*, 1991). This pilot study was aimed at verifying the existence and accuracy of the waste in place and gas flow rate data. The results of this pilot study were sufficiently encouraging such that a large-scale field study was conducted at 30 U.S. landfills. The study and its findings are described in a report entitled: "Development of an Empirical Model of Methane Emissions from Landfills," (Peer *et al.*, 1992). The objective of this study was to develop statistical models of annual landfill CH_4 emissions as a function of climate, mass of buried refuse, age, waste acceptance rate, composition, and compaction, and to develop an emission factor which could be used to estimate both U.S. and global CH_4 emissions from landfills. Sites were chosen to represent a wide range of climatic conditions, as they occur in the U.S. The research concluded that the mass of waste in place showed a significant correlation with CH_4 flow rates. The effect of refuse age on gas production was also analyzed. Gas flow rates correlated most strongly with refuse age for 10 to 20 year old refuse. Although

these results were not conclusive, they suggest that the generation time for gas production is at least 20 years. This result is analogous to the generation time of 20 to 30 years with an average of 25 years suggested by Augenstein and Pacey (1990).

None of the climate variables—precipitation, average temperature and dewpoint—proved to have significant correlations with the CH_4 flow rate. Appendix D, Table 13 summarizes data from Peer *et al.* (1992). For the purpose of this report the regression analysis was reproduced for annual rainfall (x values) and average CH_4 recovery rate per unit mass (y values). A linear regression was carried out (i.e., a line was fit through the data using the least squares method), and an analysis of variance was performed. The value for R^2 according to this analysis was 0.0047, indicating 0.47% of the variability in the observed values is explained by rainfall. Therefore, no significant correlation exists between these two variables. In addition, an exponential curve [log (average CH_4 recovery rate per unit mass) versus annual rainfall] was calculated that fit the data. Results from this analysis again revealed no significant correlation between the two variables, with an R^2 value of 0.066.

To relate CH_4 flow rates from recovery projects to CH_4 generation rates two assumptions needed to be made. It was assumed that the average recovery efficiency of a gas collection system is 75 percent (adapted from Augenstein and Pacey, 1990). Furthermore, it was assumed that 10 percent of unrecovered methane is oxidized (adapted from Whalen *et al.*, 1990). Both assumptions are subject to discussion, as very limited data exist.

Because a large amount of the variability remained unexplained in the field study described above, a decision was made to try and refine the correlation between LFG flow and waste mass. A larger LFG recovery data base was produced, which included data from most U.S. LFG recovery projects. The data base described in the *1991-1992 Methane Recovery From Landfill Yearbook* (Governmental Advisory Associates, 1991) was used as a starting point. This data base contains information on 170 U.S. LFG recovery projects. The data quality is uncertain because the data are from survey results which had not been verified. In addition, the gas flow rates in this data base are often modeled instead of measured values. To develop more accurate data, the data base developed by Governmental Advisory Associates was reviewed by AEERL and dubious information was eliminated or corrected. This effort was conducted with the Solid Waste Association of North America. In this report, the new data base, which now contains data on 105 U.S. LFG recovery plants, is referred to as the "ORD Database." Pertinent data from the ORD Database are included in Appendix D. The geographical distribution of LFG recovery plants is depicted in Figure 1.

With the expanded and verified data set in the ORD Database, it became possible to generate a regression function instead of an emission factor. This report describes the development and application of a regression model with three linear segments. Appendix A presents the statistical background of the model. The description of a second, more simple model is also included in Appendix A. This model consists of a single linear regression line and can be applied when no size-specific waste in place data are available, as is the case for most countries other than the U.S.

Figure 1. Landfill gas recovery sites in the United States.

METHANE PRODUCTION FROM THE ANAEROBIC DECOMPOSITION OF SOLID WASTE

The anaerobic decomposition of organic matter, as it occurs in a landfill is a complex process which requires that several groups of microorganisms act in a synergistic manner under favorable environmental conditions. Anaerobic refuse decomposition has been reviewed in detail by Barlaz *et al.* (1990) and more detail on the microbiology of municipal waste decomposition has been reported by Barlaz *et al.* (1989a).

Three trophic groups of anaerobic bacteria must be present to produce CH_4 from biological polymers such as cellulose, hemicellulose, and protein: (1) hydrolytic and fermentative microorganisms, (2) obligate proton-reducing acetogens, and (3) methanogens (Wolfe, 1979; Zehnder *et al.*, 1982). The hydrolytic and fermentative group is responsible for the hydrolysis of biological polymers. The initial products of polymer hydrolysis are soluble sugars, amino acids, long-chain carboxylic acids, and glycerol. Following polymer hydrolysis, the hydrolytic and fermentative microorganisms ferment the initial products of decomposition into short-chain carboxylic acids, alcohols, carbon dioxide (CO_2), and hydrogen. Acetate, a direct precursor of CH_4, is also formed.

The second group of bacteria, 'obligate proton-reducing acetogens,' convert the fermentation products of the hydrolytic and fermentative microorganisms to CO_2, hydrogen, and acetic acid. The conversion of fermentation intermediates, such as butyrate, propionate, and ethanol is thermodynamically favorable only at very low hydrogen concentrations. Thus, these substrates are utilized only when the obligate proton-reducing acetogenic bacteria can function in syntrophic association with hydrogen scavengers, such as CH_4-producing or sulfate-reducing organisms. The third group of bacteria necessary for the production of CH_4 are the methanogens. Major substrates utilized by methanogens for the production of CH_4 are acetate, formate, methanol, methylamines, and hydrogen plus CO_2 (Wolin and Miller, 1985).

While CH_4 and CO_2 are the terminal products of anaerobic decomposition, CO_2 and water are the terminal products of aerobic decomposition. Aerobic decomposition occurs in management facilities where waste is exposed to air, such as when compost is turned for aerating, and in uncontrolled dumps, such as when refuse is spread in thin layers or otherwise exposed to oxygen (*e.g.*, by scavenging). When refuse is buried in large piles, whether at an open dump or in a sanitary landfill, the oxygen entrained at burial is consumed rapidly, and substantial quantities of CH_4 may be produced (Bhide *et al.*, 1990).

Landfilled waste contains numerous constituents that have the potential to biodegrade under anaerobic conditions. The traditional method of classifying municipal solid waste (MSW) according to sortable categories [*e.g.*, paper, plastic, food waste, yard waste, glass, metals, rubber, wood, textiles, dirt, and miscellaneous (U.S. EPA, 1990)] is appropriate for recycling studies and overall solid waste management planning. However, data specific to the chemical composition of refuse are more applicable to analysis of refuse decomposition. Refuse representative of typical MSW from Madison, WI, in 1987 was reported to contain 51.2 percent cellulose, 11.9 percent hemicellulose, no more than 4.2 percent protein, and 15.2 percent lignin (Barlaz, 1988). Measurements of the cellulose concentration of Madison refuse taken from the period of 1984 through 1986 showed

values of 40 to 48 percent (Barlaz, 1985). Cellulose plus hemicellulose accounted for 91 percent of the CH_4 potential of refuse (Barlaz *et al.*, 1989b).

The components of MSW that contain significant biodegradable fractions are food waste, yard waste, and paper. Paper has a combined cellulose and hemicellulose content of 50 to 100 percent. Lignin is the other major organic component of refuse; however, lignin does not decompose significantly under anaerobic conditions (Young and Frazer, 1987).

Methane formation does not occur immediately after refuse is placed in a landfill or dump. It can take months or years for the proper environmental conditions and the required microbiological populations to become established. Numerous factors control decomposition, including moisture content, nutrient concentrations, presence and distribution of microorganisms, particle size, water flux, pH level, and temperature. Reviews of the effect of each of these factors on CH_4 production are provided in Barlaz *et al.*, (1990); Pohland and Harper (1986) and Halvadakis (1983).

The two factors that appear to have the most impact on CH_4 production are moisture content and pH. The effect of refuse moisture content has been summarized by Halvadakis (1983), although some of the data in the summary relate to manure and not municipal waste. The broadest data sets are those constructed by Emberton (1986) and Jenkins and Petus (1985). Emberton measured CH_4 production rates in excavated landfill samples under laboratory conditions. Jenkins and Petus sampled refuse from landfills and tested how CH_4 production was affected by the moisture content of refuse. In both studies, the CH_4 production rate exhibited an upward trend with increasing moisture content, despite differences in refuse density, age, and composition. It is difficult to translate the results of these laboratory studies to actual landfills. An attempt by AEERL to identify a statistically significant correlation between landfill gas recovery and annual precipitation found no such correlation, (Peer *et al.*, 1992).

A second key factor influencing the rate and onset of CH_4 production is pH. The optimum pH level for activity by methanogenic bacteria is between 6.8 and 7.4. Methane production rates decrease sharply with pH values below about 6.5 (Zehnder, 1982). When refuse is buried in landfills, there is often a rapid accumulation of carboxylic acids; this results in a temporary pH decrease and a long time-lapse between refuse burial and onset of CH_4 production which can range from months to years.

Neutralizing leachate and recycling it back through refuse has been shown to enhance the onset and rate of CH_4 production in laboratory studies (Pohland, 1975; Buivid, 1981; Barlaz *et al.*, 1987, 1989a). Given that moisture and pH have been reported as the two most significant factors limiting CH_4 production, the stimulatory effect of leachate neutralization and recycling is logical. Neutralization of leachate provides a means of externally raising the pH of the refuse ecosystem. Recycling neutralized leachate back through a landfill increases and stabilizes refuse moisture content and substrate availability. It also enhances mixing in what would otherwise be an immobilized batch reactor. Field experience with leachate recycling systems is limited and more information is needed to fully document its value. It is expected that new information will become available in the next few years.

The lapsed time preceding the onset of CH_4 production in landfills is important when considering the management of individual landfills for biogas recovery or emissions mitigation. The age at which landfills and uncontrolled dumps begin to produce CH_4 is of lesser importance when evaluating global CH_4 emissions from MSW management systems. For estimating global emissions, it is the total CH_4 production potential that is more critical.

METHANE POTENTIAL OF MUNICIPAL SOLID WASTE IN LANDFILLS

Methane potential of landfilled refuse can be determined in three basic ways. The theoretical CH_4 potential of the main chemical constituents may be calculated or laboratory tests may be conducted, imitating reality in various degrees. Also field tests have been performed. All methods have in common the question whether the data are representative or not because, even in field tests, waste composition and other parameters that affect CH_4 generation, may show unpredictable variety from one location to the next.

Knowledge of the chemical composition of refuse buried in a landfill makes it possible to estimate the maximum volume of CH_4 that may be produced. The mass of CH_4 that would be produced if all of a given constituent were converted to CH_4, CO_2, and ammonia may be calculated from Equation 1 (Parkin and Owen, 1986).

$$C_nH_aO_bN_c + [n-(a/4)-(b/2)+3(c/4)]H_2O \rightarrow [(n/2)-(a/8)+(b/4)+3(c/8)]CO_2+[(n/2)+(a/8)-(b/4)-3(c/8)]CH_4+cNH_3 \quad (1)$$

Using this stoichiometry, potential CH_4 production volume from cellulose ($C_6H_{10}O_5$) and hemicellulose ($C_5H_8O_4$) is 415 and 424 liters per dry kilogram(l/kg) at standard temperature and pressure, respectively. These methane potentials represent maximum CH_4 production if 100 percent of the cellulose and hemicellulose were converted to CH_4. However, decomposition of these constituents in landfills is well below 100 percent for several reasons, but mainly because (1) some cellulose and hemicellulose is surrounded by lignin or other recalcitrant materials (such as plastic) and, therefore, is not biologically available; and (2) without active intervention, buried refuse is not evenly exposed to moisture, microorganisms, and nutrients. Barlaz *et al.* (1989b) applied mass balances to shredded refuse incubated in laboratory-scale lysimeters with leachate recycle. Carbon recoveries of 87 to 111 percent were obtained, where a perfect mass balance would give a carbon recovery of 100 percent. Greater than 100 percent recoveries were obtained in some cases due to sampling and analytical error. Mineralization of 71 percent of the cellulose and 77 percent of the hemicellulose was measured in a container sampled after 111 days. Mass balances were useful for documenting the decomposition of specific chemical constituents and demonstrating the relationship between cellulose and hemicellulose decomposition and CH_4 production.

Stoichiometry may also be used to estimate the CH_4 potential remaining in a landfill by sampling the refuse, performing the appropriate chemical analyses, and calculating the CH_4 potential. Ideally, the initial chemical composition and CH_4 potential of the refuse

would be known, in which case comparing that initial CH_4 potential with the potential at the time of sampling would provide information on the fraction of the refuse that has been degraded. Indisputably, representative sampling of a full-scale sanitary landfill is not realistic. Sampling size is limited to volumes that can be reasonably handled and reduced by proven techniques. However, it is possible to obtain multiple samples at presumably representative locations within a landfill to get an estimate of the range and extent of decomposition.

Another technique for assessing the CH_4 potential of refuse is the biochemical methane potential (BMP) test (Shelton and Tiedje, 1984; Bogner, 1990). In the BMP test, the anaerobic biodegradability of a small sample of refuse (5 to 10 g) is measured in a small batch reactor [100 to 200 milliliter (ml)]. While the BMP represents an upper bound of CH_4 potential from refuse, it will be lower than the stoichiometric estimate described above. BMP's also require representative sampling in landfills. A recent application of the BMP test was presented by Wang et al. (1994).

Comparison of CH_4 production data between field-scale landfills and laboratory experiments is difficult because there are essentially no data in the open literature on CH_4 production rates in field-scale facilities. Interpretation of data from field-scale landfills is complicated by questions regarding the mass of refuse responsible for production of a measured volume of gas and the efficiency of gas collection. While laboratory data are of higher quality due to the more closely controlled conditions, they are not completely representative of the field. Also, data are not perfectly comparable in that experimental conditions (*e.g.*, moisture, particle size, temperature, etc.) are not uniform between studies. In addition, most laboratory experiments were conducted to explore techniques for enhancing CH_4 production. The enhanced CH_4 production rates would not be expected at field-scale landfills unless certain enhancement techniques are employed.

Methane yields of 42 to 120 l/kg dry refuse have been reported in laboratory tests conducted with leachate recycling and neutralization (Barlaz *et al.*, 1987; Barlaz, 1988; Kinman, 1987 and Buivid, 1981). These studies show significant variation in CH_4 production rate and CH_4 yield. Some of the differences can be explained by differences in experimental design. For example, the data reported by Barlaz *et al.* (1987) and Barlaz (1988) differ in reactor volume (100 vs. 2 l), temperature (25°C vs. 41°C), and the rate of leachate recycling. Also, Buivid (1981) used refuse with an abnormally high paper content.

Methane yields were measured in field-scale test cells as part of the Controlled Landfill Project in Mountain View, California (Pacey, 1989). Yields of 38.6 to 92.2 l CH_4 /dry kg of refuse were measured after 1,597 days. However, mass balance data suggested that significant volumes of CH_4 were not measured in certain test cells. A number often used by the LFG industry as an estimate of CH_4 production in field-scale landfills is 0.1 cubic feet CH_4 per wet pound per year (ft^3/wet lb-yr). Assuming refuse buried at 20 percent moisture and a 15 year period for gas production, this converts to a yield of 7.8 l CH_4/dry kg, a number comparable to some of the lower values reported in the literature.

Even in landfills with venting systems, some of the CH_4 is likely to escape from the landfill through the final cover. The fraction released through the final cover will be a function of the type of gas venting system in place and the type of cover. Probably not all the CH_4 that escapes from landfills is released to the atmosphere. Some may be converted to CO_2 as it passes through the cover soil by aerobic methanotrophic bacteria, CH_4 oxidation has been documented in landfill cover soil studied under laboratory conditions (Whalen *et al.*, 1990). However, there are no data on the quantitative significance of CH_4 oxidation above field-scale landfills. Methane escaping through cracks in a landfill cover most likely will not reside in the cover for a period sufficient to undergo significant oxidation.

MODEL TO ESTIMATE MUNICIPAL LANDFILL METHANE EMISSIONS

REGRESSION MODEL

The ORD Database includes data on LFG recovery flow rates and welled waste from 105 LFG recovery sites. Welled waste is defined as the quantity of waste from which LFG is extracted through the recovery wells. As mentioned in the Introduction, data in the ORD Database are based on measurements and estimates from on-site personnel at landfills with LFG recovery systems and were quality assured in telephone-interviews conducted by AEERL staff.

To develop a model relating flow rates to welled waste, the ORD Database was subjected to regression analysis. The objective was to let statistical criteria dictate the shape and position of the regression curve. The only constraint was that the line had to start in the origin. This resulted in a regression model with three different linear segments, where each segment applies to a distinct landfill size class. Figure 2 shows the regression curve, as well as the gas flow rate plotted against welled waste for the 105 sites from the ORD Database.[1] The analysis is described in detail in Appendix A.

The size classes and equations for the three segments of the curve are:

I	$x < 1.128$	$y = 19.822\,x$
II	$1.128 \leq x < 4.082$	$y = 1.652\,x + 20.495$
III	$x \geq 4.082$	$y = 9.195\,x - 10.294$

where: x = welled waste, (tg).
y = LFG flow rate, (m^3/minute).

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[1] Common sense would suggest that the curve is more likely to be smooth. However, due to limitations inherent to the regression analysis the curve is segmented. The position, inclination, and length of the segments are dictated by the data. Therefore, the fact that the second segment is flatter than the first and third segment may not be interpreted as a CH_4 rate drop for a certain size landfill. This second segment merely connects the two other segments in such a way that the combination of all three segments best represents the data.

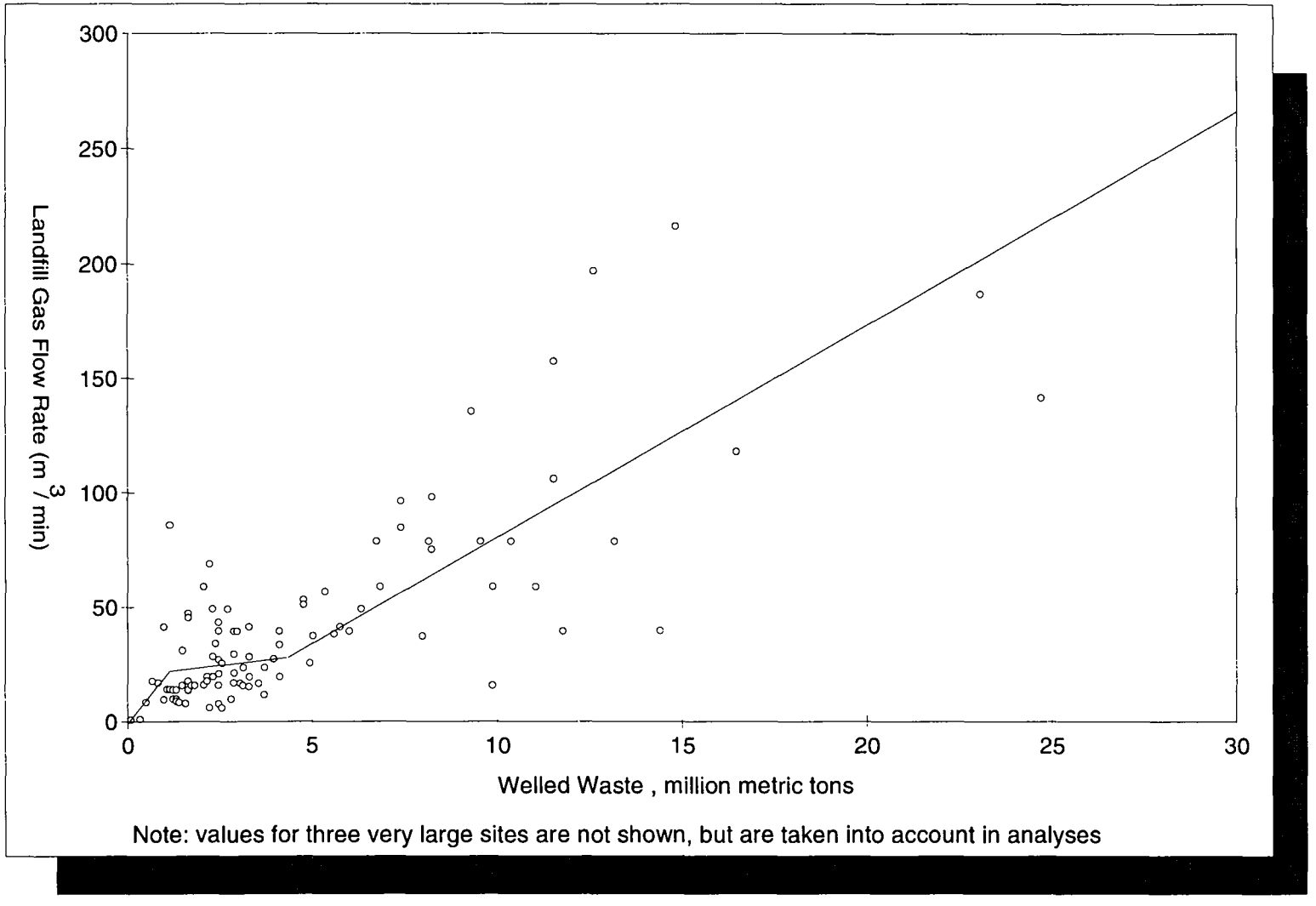

Figure 2. Landfill gas flow rates versus welled waste for 105 recovery sites and best fitting regression curve.

CONVERSION FACTOR

In order to convert LFG flow rate (y) in m^3/min to the actual mass flow of CH_4 released to the environment (Y) in g/min, the following three steps must be taken:

1. Convert the initial LFG flow, expressed in units of volume per minute, into mass CH_4 flow (Y_w). Use the relative concentration of CH_4 in LFG, c = 0.50 [Peer *et al.* (1992); Roqueta, (1992) and Anderson, (1992)], and the density ρ = 677 g/m^3. The density calculation is presented in Appendix C.

$$Y_w = y * c * \rho \quad [\frac{m^3}{\text{min}} * \frac{m^3}{m^3} * \frac{g}{m^3}] \tag{2}$$

2. Adjust for the efficiency (r) of the gas recovery system. This step actually converts the CH_4 flow rate to CH_4 generation rate (Y_g). A recovery efficiency of 75 percent is assumed (Augenstein and Pacey, 1990). Subsequently, an adjustment needs to be made for the fraction of CH_4 that does not reach the atmosphere because the CH_4 is oxidized on its way out of the landfill. This adjustment converts CH_4 generation into CH_4 emission. The oxidation factor, (o) is estimated to be 0.10 (Mancinelli and McKay, 1985).

$$Y_g = y * c * \rho * \frac{(1-o)}{r} \quad [\frac{g}{\text{min}}] \tag{3}$$

3. By introducing a factor 525,600 to convert from minutes to years, the actual mass of CH_4 released annually (M) becomes:

$$Y = y * c * \rho * \frac{(1-o)}{r} * 525{,}600 \quad [\frac{g}{\text{min}} * \frac{\text{min}}{yr}] \tag{4}$$

By employing the presented values for c, o, ρ, and r, the conversion factor is:

$$CF = 213 * 10^6 \quad [\frac{g\,\text{min}}{yr\,m^3}] \tag{5}$$

The accuracy of this number is addressed on page 18 in the section entitled: "Development of Ranges."

ESTIMATE OF TOTAL WASTE IN PLACE

MUNICIPAL LANDFILLS

In 1988, the EPA's Office of Solid Waste (OSW) published the results of a survey in which detailed information on 1,176 U.S landfill facilities was collected in a data base. The results of this survey, the Subtitle D Municipal Landfill Survey, are described in "National Survey of Solid Waste (Municipal) Landfill Facilities," (U.S. EPA, 1988) henceforth referred to as "the Survey." The Survey includes site-specific waste in place data, which can be used in the regression model described in Section 2.

The Survey was conducted in response to the Hazardous and Solid Waste Amendments of 1984. Under these amendments, EPA was required to determine whether the Resource Conservation and Recovery Act (RCRA) regulations were sufficient to protect human health and the environment from ground-water contamination. In this report, this data base will be referred to as the "OSW-Westat Database." For the Survey, a landfill qualified as a municipal landfill if it received primarily household refuse and commercial waste, and was not a hazardous waste facility (U.S. EPA, 1988). The target population included all municipal facilities in the U.S. and its five territories that had at least one active landfill unit as of November 1, 1986. At the time of the Survey, there were approximately 6,500 landfills in the U.S. For the sub-population of 1,176 landfills information was collected on ownership, location, operations, hydrogeology, waste characteristics, landfill unit construction, monitoring systems, and operating cost. Not all sites have complete data. For the purpose of this report, 142 sites had insufficient information for estimating waste in place and were eliminated. More information on why the 142 facilities were eliminated from this analysis is given in Section 5.

The following landfill data, which can be used to calculate waste in place, are included in the data base sub-population for 1,034 landfills.

- The year that waste was first placed in the landfill.
- The average annual quantity of waste received in tons.
- The total design capacity of the landfill in tons.
- The total remaining design capacity in tons.

Responses to the first and second items can be used to estimate waste in place according to what is referred to as the "RATE" method. The total waste in place was estimated by multiplying the length of time that the landfill has been accepting waste in years and the average annual quantity of waste received.

$$Waste\ In\ Place_{"RATE"} = (1987 - Year\ waste\ first\ placed\ in\ landfill) * (Average\ Annual\ Quantity\ of\ Waste\ Received) \quad (6)$$

It is also possible to calculate waste in place for each landfill by taking the difference of the results from the third and fourth items. This method is referred to as the DIFF method.

$$Waste\ In\ Place_{"DIFF"} = Total\ Design\ Capacity - Remaining\ Design\ Capacity \quad (7)$$

In this report, preference is given to the DIFF method. A detailed discussion and comparison of the DIFF and RATE methods is included in Section 5. Appendix A includes estimates based on both methods.

Through a stratified sample design, large facilities in the Survey had a higher probability of being sampled than small facilities. The distribution of waste in place obtained from the subset of landfills included in the municipal solid waste landfill survey was applied to the whole population of active municipal landfills. This is done through the application of scaling factors, also referred to as weights or raising factors. The eligible sample of landfills was broken down into two strata. Stratum 1 was comprised of large or active facilities that received at least 500 tons of waste per day. Stratum 2 consisted of small or less active facilities, those that received less than 500 tons of waste per day. Fifty-two percent of the eligible active facilities were surveyed, while 13 percent of the eligible, less active sites were investigated. (This is the optimal stratified sampling plan for a sample size of 1,000, in that the variance of the estimate of the total quantity of waste received is minimized.) This allocation resulted in a large, active site being roughly four times more likely to be sampled than a small facility.

To extrapolate the sub-population to the total population of U.S. landfills, as well as to account for the difference in likelihood to be sampled, the scaling factors for active and less active landfills are 2.00 and 7.00, respectively. These particular scaling factors were presented with the data base. By multiplying the scaling factor by waste in place for each site from the sub-population and adding up the results, total waste in place for the nation can be calculated. Section 5 addresses the quality of the scaling factors. Appendix A gives the statistical background for the determination of scaling factors.

According to the DIFF method, the total waste in place in the U.S. in 1986, the year the Survey was conducted, was 4,720 tg. The average landfill age was 19 years. The total average annual quantity of waste received was 248 tg. The rate at which MSW is landfilled has been declining by approximately 2% per year between 1987 and 1993 (U.S. EPA, 1992). Because this decrease is small and information on changes in disposal rates of other types of waste such as demolition and construction debris have not been found, the annual disposal rate of 248 tg/yr was assumed to be steady between 1987 and 1993. Hence, for 1992 the total waste in place for the U.S. was 6,200 tg. Estimates for the RATE method are given in Appendix A.

Magnitude of Waste Quantity

In the U.S., 248 tg/yr of waste is estimated to be placed in municipal landfills. This amount is equivalent to a daily, per capita placement rate of 3.0 kg. (6.6 lb/cap/day) [U.S. population of 226,505,500 (1990)]. It should be emphasized that this number is not equivalent to the MSW generation rates computed by OSW and published in their bi-annual "Characterization of Municipal Solid Waste" reports[2] [U.S. EPA, (1990) and U.S. EPA, (1992)]. Waste generation computed by the EPA/OSW study amounts to approximately 2.0 kg/cap/day (1992) and does not include such wastes as demolition and construction debris, nonhazardous industrial waste, sludges, fly ash, etc., all of which may (in part) be landfilled. Instead, the 3.0 kg rate generated here represents the waste placement rate which is actually landfilled in municipal landfills in the U.S. based on the Survey and OSW data base. As Table 1 indicates, this placement rate is comparable to those of other industrialized countries.

TABLE 1. COMPARISON OF WASTE GENERATION AND WASTE-TO-LANDFILL RATES FOR EIGHT INDUSTRIALIZED COUNTRIES

Waste (kg per capita per day)	MSW Generated	Total Waste to Landfill	Average % to Landfill	Landfillable Waste[1]	Source[2]
United States	2.0	3.0	70	4.3	This report
Japan	1.0	1.6	40	3.8	Cossu
United Kingdom	2.0	4.2[3]	97	4.3	Cossu
Netherlands	1.6	2.5	51	5.1[4]	Beker
Sweden	0.8	2.1	80	2.6	Nillson
Denmark	1.2	1.8	60	3.1	Christensen
Finland	1.4	24.1[5]	varies	n/a	Ettala
Italy	0.8	3.4	100	3.4	Cossu & Urbini

[1] Assumed that the average % to landfill (previous column) is 100 %.
[2] All references from "International Perspectives on Municipal Solid Wastes and Sanitary Landfilling," (Carra and Cossu, 1990).
[3] Excluding mining, agricultural, and power station wastes which amount to 16 kg/cap/day!
[4] Mainly demolition, construction, and nonhazardous industrial waste.
[5] Wastes from mining and wood harvesting and processing are also landfilled.

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[2] These waste generation estimates are based on a materials flow methodology using production output figures from, for instance, the Department of Commerce.

INDUSTRIAL LANDFILLS

The U.S. maintains separate records of industrial waste that is landfilled at specially designated industrial landfills, instead of at municipal landfills. Estimates by Schroeder *et al.*, (1987) indicate that 15 tg of biodegradable industrial waste is landfilled annually at industrial landfills. In this report it is assumed that the size as well as the age distribution of industrial landfills is similar to that of municipal landfills. Because the average age of a landfill is 19 years, the total amount of landfilled industrial waste is 285 tg. For 1992, this number would be 375 tg, which is 15 tg multiplied by 25 years.

ESTIMATE OF U.S. LANDFILL METHANE EMISSIONS

The ORD Database, which presents data from 105 LFG recovery sites, was used to generate a model which describes CH_4 gas flow rate as a function of refuse mass in place. In order to use flow rates as a surrogate for CH_4 emissions, certain assumptions needed to be made. These assumptions are expressed in a conversion factor. This factor accounts for the recovery efficiency of LFG projects and for the probable loss of CH_4 due to oxidation. Furthermore, the factor incorporates a conversion from LFG flow rate (m^3/min) to actual mass CH_4 emissions. Application of the model to the waste mass data from the OSW-Westat Database and multiplication of the results with the conversion factor yields national CH_4 emissions from landfills.

Since the developed regression model is tripartite, CH_4 emissions for each landfill depend on size classification. Therefore, it is not possible to calculate the total nation-wide waste in place first and then apply the model. Instead, the proper approach is to divide the landfill population into the three size classes. For each site the LFG flow rate is calculated by applying the appropriate regression equation to the waste in place. This rate is in turn multiplied by the appropriate scaling factor for each site. The estimated values of LFG flow rates are then added together, yielding an estimate of total LFG flow rate for all active municipal landfills. Ultimately, the total LFG flow rate is multiplied by the conversion factor to convert LFG flow rate to mass of emitted CH_4 in grams per year. This estimate is then adjusted for the amount of CH_4 being recovered or flared. In 1992, 1.2 tg of landfill methane was utilized (Thorneloe, 1992a). There are also landfills that flare the gas instead of recovering the energy. As no quantitative data are available, the amount of LFG that is flared is assumed to be 0.5 tg/yr (i.e. 40 percent of LFG recovered for energy purposes). Therefore, the total amount of recovered CH_4 for 1992 is estimated to be 1.7 tg/yr.

It is estimated that an additional 15 tg of industrial waste is landfilled annually in the U.S. at industrial landfill sites. Because limited information on industrial landfills is available, it is assumed that industrial landfill characteristics, including size and age distribution, are similar to those of municipal landfills. The average age of an industrial landfill is then also assumed to be 19 years, equal to the age of a municipal landfill. Then the total amount of landfilled industrial waste is 285 tg. The 285 tg was apportioned to the three size classes used to classify municipal landfills. Tables 2 and 3 illustrate the calculations for CH_4 emissions from waste in industrial landfills.

Tables 4, 5 and 6 summarize the waste in place data as well as the nationwide emissions for 1986 and 1992. In Tables 5 and 6 emissions are based on waste calculations by the DIFF method. Emissions estimates making use of waste in place calculated by the RATE method may be found in Appendix A. For 1986 CH_4 emissions from U.S. landfills were estimated at 11 tg/yr with lower- and upper bound values of 7 and 15 tg/yr respectively. Methane emissions from U.S. landfills in 1992 were estimated at 13 tg/yr with lower and upper bound values of 9 to 19 tg/yr. Appendix B compares these estimates with other previously published estimates. The development of the lower and upper bound values is discussed on page 18 in the section entitled: "Development of Ranges."

TABLE 2. METHANE EMISSIONS FROM INDUSTRIAL LANDFILLS (Metric Units)

Size Class		I	II	III	Total
Industrial Waste in Place, (tg)	1986	17	34	234	285
	1992	23	45	307	375
LFG Flow Rate, (m^3/min)	1986	339	77	2,138	--
	1992	455	95	2,813	--
Conversion Factor, (10^6 g.min/yr.m^3)		213			
Emissions Estimate, (tg/yr)	1986	0.07	0.02	0.46	0.55
	1992	0.10	0.02	0.60	0.72

Note: Mass industrial waste was calculated by the DIFF method, apportioning the total industrial waste in place to the size classes, proportionate to the fraction of municipal waste in each size class.

TABLE 3. METHANE EMISSIONS FROM INDUSTRIAL LANDFILLS (U.S. Units)

Size Class		I	II	III	Total
Industrial Waste in Place, (10^9 tons)	1986	0.019	0.037	0.258	0.314
	1992	0.025	0.05	0.338	0.413
LFG flow rate, (CFM)	1986	11,934	2,727	7,5519	--
	1992	16,065	3,348	99,333	--
Conversion Factor, (tons.min/yr. cu. ft)		4,860			
Emissions Estimate, (10^6 tons/yr)	1986	0.08	0.02	0.50	0.60
	1992	0.11	0.02	0.66	0.79

Note: Mass industrial waste was calculated by the DIFF method, apportioning the total industrial waste in place to the size classes, proportionate to the fraction of municipal waste in each size class.

TABLE 4. TOTAL WASTE IN PLACE AND TOTAL METHANE EMISSIONS FROM MUNICIPAL LANDFILLS PER SIZE CLASS (Metric Units) (Not corrected for industrial landfills, flaring or utilization)

Size Class		I	II	III	Total
Size Class Boundaries, (tg)		$x < 1.13$	$1.13 \leq x < 4.08$	$x \geq 4.08$	
Number of Sites	1986	816	144	74	1,034
	1992	781	154	99	1,034
Percent of Sites, (%)	1986	79	14	7	100
	1992	76	15	10	100
Municipal Waste in Place, (tg)	1986	898	1,169	2,639	4,720
	1992	942	1,452	3,797	6,200
Percent of Waste in Place, (%)	1986	19	25	56	100
	1992	15	23	61	100
LFG Flow Rate, (10^4 m^3/min)	1986	1.8	1.3	2.2	5.3
	1992	1.9	1.6	3.2	6.7
Conversion Factor, (10^6 g.min/yr.m^3)		213			
Emissions Estimate, (tg/yr)	1986	3.8	2.8	4.7	11.3
	1992	4.1	3.4	6.8	14.2

DEVELOPMENT OF RANGES

Due to the impossibility of estimating errors associated with the assumptions used to estimate CH_4 emissions from landfills a mathematical approach in which individual errors are propagated is meaningless. Therefore, the following method was adopted. The standard deviation in the emissions estimate is 12 percent (Appendix A). Approximate 95 percent confidence intervals are obtained by adding plus/minus two standard deviations to the estimate. Consequently, it is assumed that errors in other parameters amount to at least another 12 percent. Therefore, ranges are expressed by adding plus/minus 36 percent to the emissions estimate.

TABLE 5. TOTAL METHANE EMISSIONS FROM U.S. LANDFILLS (Metric Units)

		1986	1992
Total Municipal Waste in Place, (tg)		4,720	6,200
LFG flow rate, (10^4 m^3/min)		5.3	6.6
Estimated Emissions from Municipal Landfills, (tg/yr)		11.2	14.2
Emissions from Industrial Landfills, (tg/yr)		0.55	0.72
Methane Currently Recovered or Flared, (tg/yr)[1]		1.2[2] (1.1)	1.7 (1.5)
Estimated Total U.S. Emissions, (tg/yr)	Lower bound	7	9
	Mid-point	11	13
	Upper bound	15	18

[1] Amount of recovered CH_4 needs to be adjusted for the fact that it would have undergone oxidation if it had not been recovered. So it needs to be multiplied by 100 - 10 = 90 %. Adjusted amount is in parentheses.

[2] Amount of CH_4 recovered in 1986 assumed to be 70% of 1992 value.

TABLE 6. TOTAL METHANE EMISSIONS FROM U.S. LANDFILLS (U.S. Units)

		1986	1992
Total Municipal Waste in Place, (10^6 tons)		5,200	6,830
LFG flow rate, (10^4 CFM)		187	233
Estimated Emissions from Municipal Landfills, (10^6 tons/yr)		12.5	15.7
Emissions from Industrial Landfills, (10^6 tons/yr)		0.60	0.79
Methane Currently Recovered or Flared, (10^6 tons/yr)[1]		1.3[2] (1.2)	1.9 (1.7)
Estimated Total U.S. Emissions, (10^6 tons/yr)	Lower bound	8	10
	Mid-point	12	15
	Upper bound	16	20

[1] Amount of recovered CH_4 needs to be adjusted for the fact that it would have undergone oxidation if it had not been recovered. So it needs to be multiplied by 100 - 10 = 90 %. Adjusted amount is in parentheses.

[2] Amount of CH_4 recovered in 1986 assumed to be 70% of 1992 value.

UNCERTAINTIES

This section discusses areas of uncertainty in the estimate of CH_4 emissions from landfills. Specific uncertainties regarding the statistics applied in the calculations of waste in place and CH_4 emissions can be found in Appendix A.

QUALITY OF OSW-WESTAT DATABASE

Data from the 1986 Subtitle D Municipal Landfill Survey are included in the OSW-Westat Database and the accompanying Survey document published in 1988. Although this document is quite comprehensive and explicit, there are questions regarding whether it is reasonable to extend the data base to 1992-1993. It appears that changes were made to the data base which have not been documented sufficiently. For example, no documentation was found that included the density used to convert from volume to mass for landfilled waste in the OSW-Westat Database. Also, the Survey document reported the total annual quantity of waste received as 208.8 million tons, whereas the quantity of waste received now totals 273 million tons (248 tg). Geswein (1993) stated that this difference was caused by an error in the 1986 data base, which was corrected after publication. Geswein (1993) indicated that the 248 tg/yr estimate is the most accurate. A number of the same magnitude was also used by EPA's Office of Air Quality Planning and Standards in preparation of the new regulation for new and existing landfills under the Clean Air Act.

Comparison of RATE and DIFF Methods

This section presents a comparison of the DIFF and RATE methods and summarizes the advantages and disadvantages of each method. Conclusions drawn from a numerical comparison of the DIFF and RATE methods seem to confirm the doubt regarding the quality of the OSW-Westat Database.

Waste in place can be approximated using two different methods. The RATE method uses information on the length of time that the landfill has been accepting waste (age of landfill) and the average annual quantity of waste received [Equation (6)]. According to the DIFF method, waste in place can be calculated based on the difference between total and remaining design capacity in volume units [Equation (7)].

$$Waste\ In\ Place_{RATE} = (1987 - Year\ waste\ first\ place\ in\ landfill) * (Average\ Annual\ Quantity\ of\ Waste\ Received) \quad (6)$$

$$Waste\ In\ Place_{DIFF} = Total\ Design\ Capacity - Remaining\ Design\ Capacity \quad (7)$$

In order to compare the RATE and the DIFF methods, a sub-population of 1,034 landfills was identified for which data were available to calculate waste in place using both methods. Table 7 presents the results of the comparison. The data base contains 1,175 entries. Data needed to calculate waste in place using either method are not available for 81 landfills. For 30 of the remaining landfills, no data are available to estimate waste in place using the DIFF method, and an additional 30 landfills do not have sufficient data to calculate waste in place using the RATE method. For the sub-population of 1,034 landfills, waste in place calculated by the RATE method is approximately one and a half times higher than waste in place calculated by the DIFF method. Table 8 presents summary statistics for each method.

However, a comparison between the two methods of calculating waste in place for individual landfills show significant inconsistencies, which are illustrated in Figures 3 and 4. In Figure 3, landfill sizes calculated by the DIFF method are plotted against landfill sizes calculated by the RATE method. In Figure 4, for each site, the logarithm of the quotient of waste in place estimated by the RATE method and the DIFF method is plotted against the age of the landfill. In Figure 4, the methods should present identical or at least comparable waste in place estimates, with the logarithm of the quotient being zero or close to zero. Neverthcless, for several landfills, the waste in place calculated by one method is a factor 10 or more than waste in place calculated by the other method. for only 32 percent of landfills waste in place estimates by both methods are within a 40 percent range of each other. One could possibly expect that the two methods would show different results for older landfills due to changes in compaction between waste and varying or misinterpreted acceptance rates. However, the site-specific differences in waste in place estimates also occur for relatively young landfills. For these landfills, it is unlikely that the amount of waste is estimated erroneously by either of the methods. The magnitude of the differences raises concern about the quality of the data.

From the comparison of the results of the two methods, it is not clear which of the two methods provide a more accurate estimate of waste in place. The DIFF method is based on the difference of only two parameters (Equation 7). Also it would take into account the compaction of the waste. The RATE method on the other hand, is based on a parameter (*i.e.*, the average annual quantity of waste received) which has to be calculated incorporating historical data over the life of the landfill (Equation 6). Also, it does not adjust for compaction of waste already in the landfill.

A further consideration is the uncertainties associated with waste in place estimates using the RATE method. The Survey asked respondents to report the average annual quantity of waste received over the life of the landfill. No information is available on how each respondent calculated the response to this question. In 1986, the average age of the landfills in the data base was 19 years and 170 of the landfills had been operating for 30 years or longer. Particularly the older landfills may not have kept accurate historical records on the annual amount of waste received. Also, it is unlikely that the average annual quantity of waste for these older landfills was calculated with a high degree of precision, by averaging out waste acceptance data from records (if available) spanning several decades.

TABLE 7. COMPARISON OF DIFF AND RATE METHODS FOR WASTE IN PLACE CALCULATIONS

	DIFF	RATE
Total number of landfills in the data base	1,175	1,175
Landfills with no data for calculating waste in place by either method	81	81
Landfills to which the indicated method cannot be applied	60	60
Remaining sample population	1,034	1,034
Total population (sum of scaling factors)	6,223	6,223
Waste in place for sub-population of 1,034 landfills, (10^9 tons)	1.58	2.30

TABLE 8. COMPARISON OF DIFF AND RATE METHODS FOR WASTE IN PLACE CALCULATIONS: STATISTICAL SUMMARY

	DIFF	RATE
Mean (million tons)	1.52	2.23
Standard error (million tons)	0.26	0.31
Median (million tons)	0.12	0.14
Mode (million tons)	0.23	0.18
Standard deviation (million tons)	8.55	9.79
Variance	$7.3*10^{13}$	$9.6*10^{13}$
Kurtosis	374	632
Skewness	23	17
Range (million tons)	2.44	2.44
Minimum (tons)	46	70.00
Maximum (million tons)	244	244

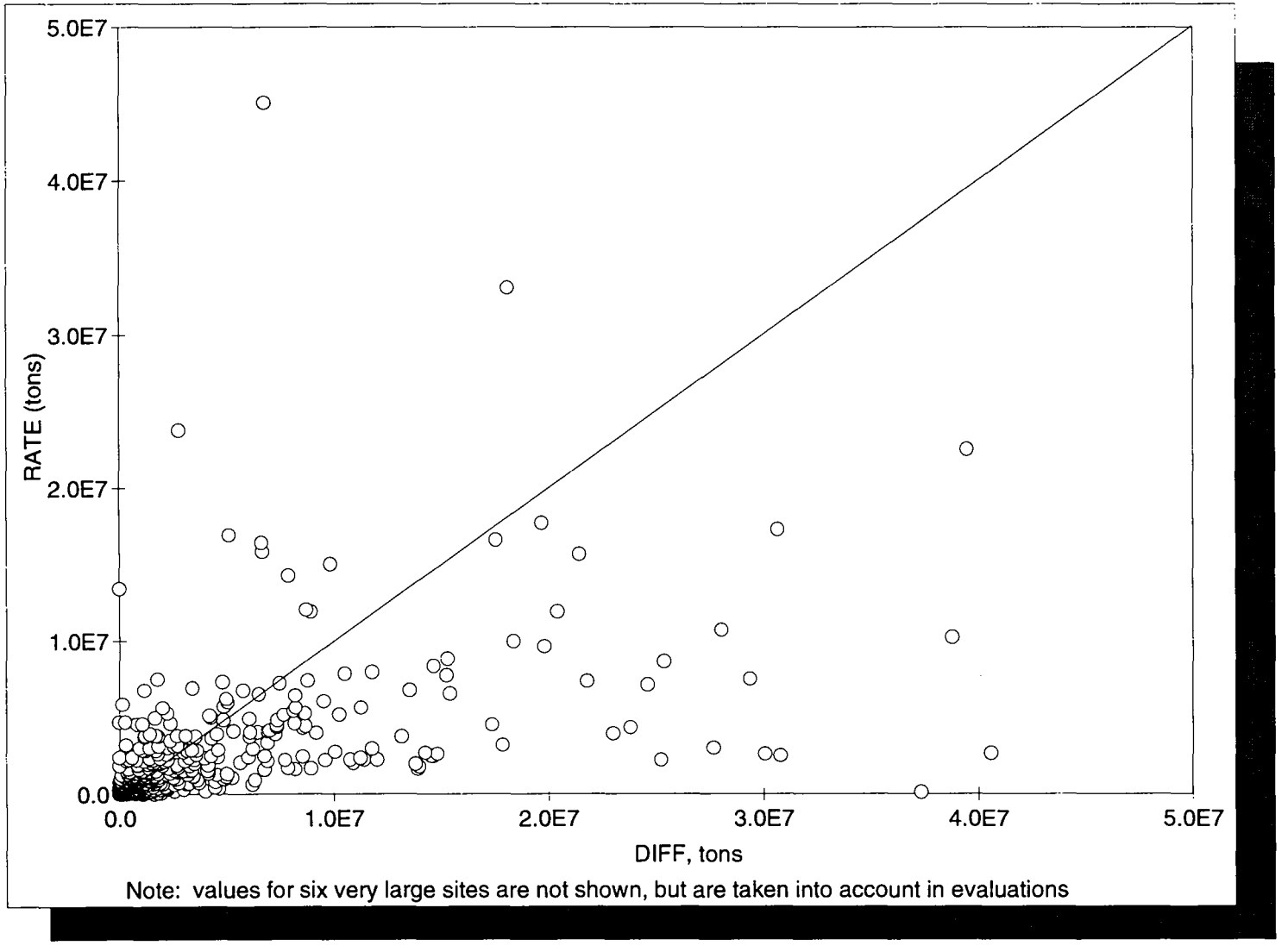

Figure 3. Comparison of waste in place data for RATE and DIFF methods for all landfills from sample population.

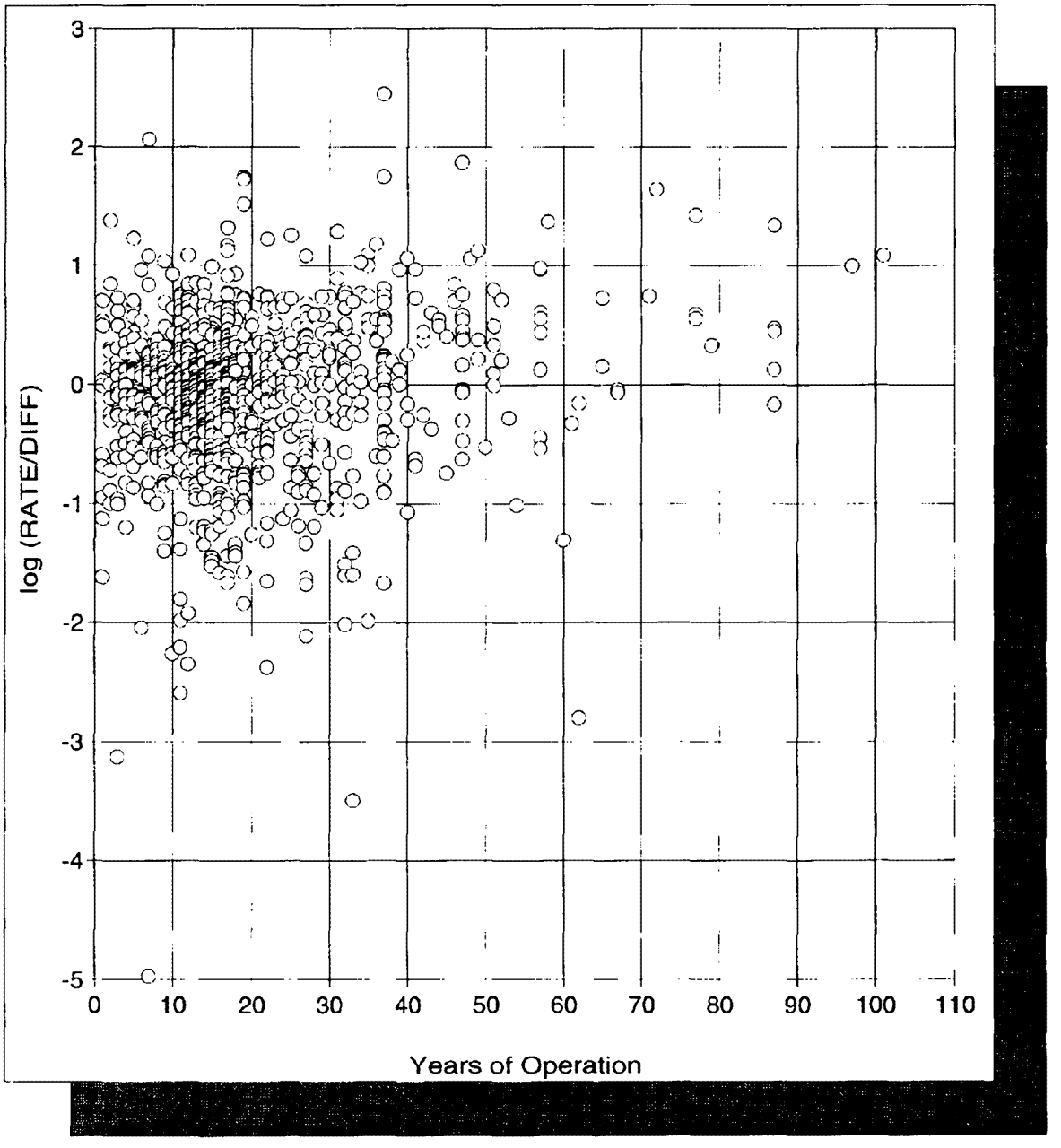

Figure 4. Comparison of quotient of waste in place data for RATE and DIFF methods versus age of landfill from sample population.

Nevertheless, it is not unreasonable to suspect that certain respondents based their estimate of the average annual quantity of waste received on the acceptance rate for the most recent year or years prior to 1986 when the survey was conducted. If this is true, the average annual acceptance rates used in the RATE method probably are too high, because national annual waste generation has increased substantially over the past decades. (This issue is discussed in Section 6, "Trends in Waste Management and Their Impact on Future Emissions").

An issue that should be raised regarding the DIFF method is how respondents were expected to come up with an estimate for the remaining capacity of the landfill. In reality, landfill owners do not usually know how much waste there is in their landfill, nor how much the remaining capacity might be. The survey questionnaire tried to circumvent this problem by providing the definition for remaining design capacity as: "total design capacity minus amount of waste currently in the landfill." If it is difficult to estimate the amount of waste in a landfill directly. The only other way of estimating the quantity of waste currently in a landfill is by recording the flow of waste going into the landfill. If this is true, respondents would have had to use some version of the RATE method for their estimates of waste in place. Although this would not explain the difference in the waste in place estimates using the two methods, it does suggest that the DIFF method data in the data base may be based on respondent estimates of total waste in place using the RATE method thus, implying that both methods may be mathematically the same.

Scaling Factors

Table 7 shows that the original data base contained information from 1,175 sites. These sites were a sub-population of the approximately 6,500 landfills in the U.S. Of the 1,175 surveyed sites, 1,102 were considered eligible for the intended purpose of determining the risks as specified under RCRA, as described in section 3. The Survey states that the 1,102 sites in the sub-population would correspond to a total of 6,034 landfills. The applied scaling factors are not provided in the Survey document. (Scaling factors merely describe the mathematical procedure to go from 1,102 to 6,034; addition of all the scaling factors yields the total population of landfills.) Instead, scaling factors were listed in the data base, having values of either 2.00 for landfills receiving at least 500 tons of waste per day and 7.00 for landfills receiving less than 500 tons of waste per day. (There are a few values of 1 and 3 which were considered erroneous and set equal to 2.00.) For lack of better information, it was assumed that the scaling factors of 2.00 and 7.00 are based on the size of the original populations of eligible and total landfills.

For the purpose of determining waste in place, it was not possible to use all 1,102 entries. Only 1,034 sites had sufficient data. Addition of the scaling factors for these 1,034 landfills corresponds to a total number of 6,223 landfills, which is 3 percent more then the original count of 6,034 landfills. Therefore, the scaling factors of 2.00 and 7.00 may be too high. Consequently, the estimate of total waste in place may also be too high. Since the original sub-population of 1,102 sites cannot be defined exactly, it is not correct to assume that total waste in place or CH_4 emissions is overestimated by the same percentage. To eliminate this error, more about the history and validity of the current scaling factors and sub-populations needs to be known.

UNCERTAINTIES ASSOCIATED WITH THE MODEL

Emissions from Small Landfills

There is anecdotal evidence that emissions per ton of waste are higher from small landfills than from larger landfills. (Small landfills are defined as landfills from Size Class I with less than 1.128 tg of waste in place). This supposition is strengthened by the fact that the slope of the first segment, belonging to Size Class I, in Figure 2 is steeper than the slopes of the other two segments. A steeper slope relates to a higher emission factor. This is detailed in Appendix A. However, the regression analysis for the first segment is based on data from only 5 LFG recovery projects and should, therefore be considered with caution.

The reason why there are only a few LFG recovery projects at smaller landfills is that the feasibility of a LFG recovery operation will generally increase with the size of the landfill. To have a feasible LFG recovery project at a small landfill, the LFG yield per ton of waste has to be higher than at an identical project at a large landfill, i.e. it has to be a "rich" landfill. In addition, a LFG recovery project at a small landfill would have to be more efficient than would be necessary at larger sites. Hence, a bias may be introduced to the data, causing the emission factor for smaller landfills to be relatively high.

Although more research would be necessary to resolve this issue, it is important to point out that small landfills, though numerous, contain only a small fraction of the total amount of refuse disposed of in the U.S. For instance, 86 percent of all active MSW landfills in the U.S. has less than 1 tg of waste in place (1986). Nevertheless this population only contains 17 percent of total waste in place.

Bias toward "Rich" Landfills

The reasoning from the previous section may be applied in broader terms. To ensure sufficient return on investment LFG developers will pick those landfills with the highest LFG potential ("rich" landfills). The methodology employed in this report extrapolates LFG flow versus welled waste data (i.e., emission factors) to include all waste at U.S. municipal landfills. Since the emission factors are only based on landfills that do have LFG projects (the "rich" ones), a bias may be introduced causing the estimates to be too high. Alternately, most LFG recovery projects are in the Northeast and in California (Figure 1). Perhaps this is because the high population density in these areas leads to bigger landfills. Also, the energy economics may be better in these geographical regions.

However, arguments are also available to suggest that the aforementioned bias may be lower than would be expected. Especially in the past when waste with high CH_4 potential may have been combined with waste with low CH_4 potential. Also, CH_4 potential assessments usually have a large margin of error, and developers may inadvertently install wells in "poor" sections or layers of the landfill. In addition, LFG feasibility studies will usually involve economics that is more complicated than straight forward profitability calculations. If a landfill owner is required to control LFG emissions it may be beneficial to choose LFG recovery, even if no profits are ever generated. Returns from gas sales would help offset part of the required investment for the otherwise

obligatory flare. In addition, there may be certain incentives or tax breaks that would make the development of marginal landfills possible.

OTHER UNCERTAINTIES

Extrapolation to 1993

According to the DIFF method, the total waste in place in the U.S. at the beginning of 1987 was 4,720 tg. Total waste in place was updated to the beginning of 1993 by linear extrapolation. Uncertainties in the waste estimates, as well as in methane emissions estimates will be magnified by this multiplication. Section 6, "Trends in Waste Management and Their Impact on Future Emissions," discusses factors which influence the update of the 1986 data from the Survey.

Industrial Waste

It is estimated that an additional 15 tg of industrial waste is landfilled annually in the U.S. in industrial landfills (Schroeder *et.al.,* 1987). Compared to the annual waste disposal rate of 248 tg/yr at municipal landfills this is only a small amount; 6 percent. Very limited information on industrial waste, other non-MSW, and on industrial landfills is available. It is assumed that industrial landfill characteristics, including size and age distribution, are similar to those of municipal landfills. To reduce uncertainty, more reliable data for industrial landfills would have to be obtained.

Generation Time

The generation time is the "life time" of a batch of waste during which it produces CH_4. Typically, a generation time of 20 to 30 years is assumed to be reasonable for temperate climates (Augenstein and Pacey, 1990; EMCON Associates, 1982). The calculations in this report do not explicitly consider the generation time. The reason is that the regression model is developed from data from LFG recovery projects at landfills which have waste of different ages. Therefore, the regression model already accounts for generation time and differences in waste age. However, this holds true only if the age distribution of waste in landfills in the U.S. was accurately represented by the age distribution of landfills used to develop the regression model.

Variables Used in the Conversion Factor Derivation

The MASS of waste in place is expressed in metric tons. Data on mass of refuse are a large source of error as they are gathered by site operators and are not always properly documented. Mass of refuse can be calculated by several different methods. At a few sites the trucks are weighed at the gates (U.S. EPA, 1992). In most cases, however, the operators keep count of the number of trucks and estimate the load. Refuse density in the U.S., has been estimated to range from 500 to 1,300 lbs/yard^3 (300 to 750 kg/m^3) depending on the characteristics, compaction, and humidity of the waste.

The GAS FLOW is measured in m^3/min or ft^3/min. Gas flow data need to be defined at a certain standard pressure and temperature. Within the EPA, a standard pressure of 1 atmosphere (atm) and a standard temperature of 20°C (68°F) are often used. Nevertheless, there is reason to adopt a standard temperature of 16°C (60°F), as this is the temperature used throughout the LFG industry as well as the petroleum and natural gas industries (Roqueta, 1992; Anderson, 1992 and American Gas Association, 1985). The inaccuracy of flow meter readings plays a minor role in overall uncertainty of landfill emissions compared to other parameters used in estimating CH_4 emissions from landfills. One source of error can be found in the conversion of actual gas flows to dry standard conditions. This conversion is accounted for in the density calculation.

The DENSITY of CH_4 at 0°C and 1 atm was calculated assuming that CH_4 is an ideal gas. The density of CH_4 at 1 atm and 60°F (16°C) is 677 g/m^3. This calculation is demonstrated in Appendix C.

The average relative CONCENTRATION of CH_4 in LFG for 21 field study sites was 50.1 percent while the averages for the individual landfills ranged from 40.2 to 58.1 percent (Peer *et al.*, 1992; Roqueta, 1992 and Anderson, 1992). In this report, a concentration of 50 percent was assumed (factor = 0.50).

The OXIDATION of CH_4 is estimated to be 10 percent, [*i.e.*, $o = 0.10$ (Mancinelli and McKay, 1985)]. Whalen (1990) found a large range of oxidation rates, leading him to estimate that as much as 50 percent of landfill CH_4 might be oxidized before it reaches the surface. Due to the common occurrence of cracks and fissures in the landfill surfaces, which would reduce contact between oxidizing organisms and the LFG, the lower value of 10 percent is used. More research is needed to determine the effect of oxidation on potential emissions.

The RECOVERY EFFICIENCY of LFG projects is thought to be highest when the projects are undertaken to comply with regulatory programs; however, this cannot be assumed in all cases. The overall recovery efficiency is typically affected by well spacing and the presence or permeability of the cover layer. The only published estimate of gas recovery efficiency is based on expert judgements and gives a most probable value of 75 percent with lower and upper bounds of 50 to 90 percent (Augenstein and Pacey, 1990).

TRENDS IN WASTE MANAGEMENT AND THEIR IMPACT ON FUTURE EMISSIONS

BACKGROUND

Landfill practices in the U.S. are undergoing changes that will affect LFG emissions and, consequently, CH_4 emissions from landfills. Future landfill CH_4 emissions will be influenced by several factors:

- Amount of waste landfilled, which depends on:
  - Population growth.
  - Per capita waste generation, which depends in part on economic growth.
  - Increased public awareness of the consequences and hazards of waste generation and disposal.
  - Regulatory requirements affecting waste landfill practices.
- CH_4 recovery, which depends on:
  - Regulatory requirements for controlling emissions from landfills.
  - The price of energy.
  - Measures to promote the use of CH_4 from landfills.
  - Availability of more cost-effective CH_4 recovery technologies.

The influence of these factors will result in a number of changes, including:

- Changes in solid waste generation (e.g. resulting from source reduction).
- Changes in solid waste composition.
- Increased recycling, composting, and other methods of waste treatment.
- Changes in landfill waste management.
- Increased control or recovery of LFG as a result of regulatory requirements and evolution of advanced technologies.

The trends resulting from these changes are discussed in more detail in the following text.

TRENDS IN SOLID WASTE GENERATION, COMPOSITION, AND MANAGEMENT

Sewage sludge and distinct types of industrial waste contain degradable organic carbon fractions that may contribute to LFG generation. Other types of non-MSW, for instance, demolition and construction debris, fly ash, and also wood contain no or very little degradable organic carbon. Based on data from other countries (Carra and Cossu, 1990) it seems that wastes with little or no organic carbon make up the majority of the non-MSW stream. It is therefore assumed that MSW is the prime contributor to CH_4 generation. In addition, no records on the generation and disposal of non-MSW in the U.S. were found. Therefore, this section focusses entirely on trends affecting MSW generation and disposal.

Between 1960 and 1990, generation of MSW grew steadily, from 88 million tons to over 195 million tons per year. Over this period, the per capita generation of MSW increased from 2.7 pounds to 4.5 pounds per day. OSW has made projections for the year 2000, which indicate that per capita generation of MSW is expected to increase, but at a substantially slower rate compared to 1992. The projection for the amount of MSW that will be generated by the year 2000 is 222 million tons per year (U.S. EPA, 1992). MSW generation is difficult to predict because it depends on various factors including consumer preference which can change in response to social trends. For example, consumer preferences can affect the demand for lighter packaging materials or, conversely, more durable goods. Other factors which can affect the amount of MSW generated include increased consumer awareness of environmental issues and new regulations.

Recycling has been increasing over the last three decades. In 1992, 17 percent of generated MSW was recovered for recycling and composting. It is expected that this percentage will continue to rise. Projected scenarios indicate that between 25 and 35 percent of MSW will be recycled by 2000. Another alternative to landfilling of MSW is incineration. In 1990, 16 percent of MSW was incinerated. The fraction of MSW that will be incinerated in the year 2000 is estimated to increase to 21 percent (U.S. EPA, 1992a), although this increase would be contingent upon public acceptance, which has been an issue recently. Due to trends in recovery (recycling and composting) and incineration, the amount of MSW that is landfilled is projected to decrease 16 percent from 130 tons per year in 1990 to an estimated 109 tons per year in 2000 (U.S. EPA, 1992a). Figure 5 shows MSW generation and waste management trends in the U.S. from 1960 to 2000.

The difficulty in siting new landfills will probably lead to fewer and generally larger landfills than exist today. Due to increasing regulatory costs, many less efficient landfill operators may be forced to close reducing the number of landfills even further. This trend is illustrated by the annual acceptance and remaining design capacity data for landfills in the OSW-Westat Database from the Survey. Assuming a constant annual waste acceptance rate, these data suggest that by the year 2000, approximately 38 percent of currently operational landfills will have closed. The trend toward fewer but larger landfills should be favorable for the LFG recovery industry, because the feasibility of gas recovery projects tends to increase with the amount of waste in place. Stricter landfill design regulations may also benefit the CH_4 recovery industry; for example, compulsory liners would increase CH_4 recovery efficiency if the liners reduce lateral gas leakage.

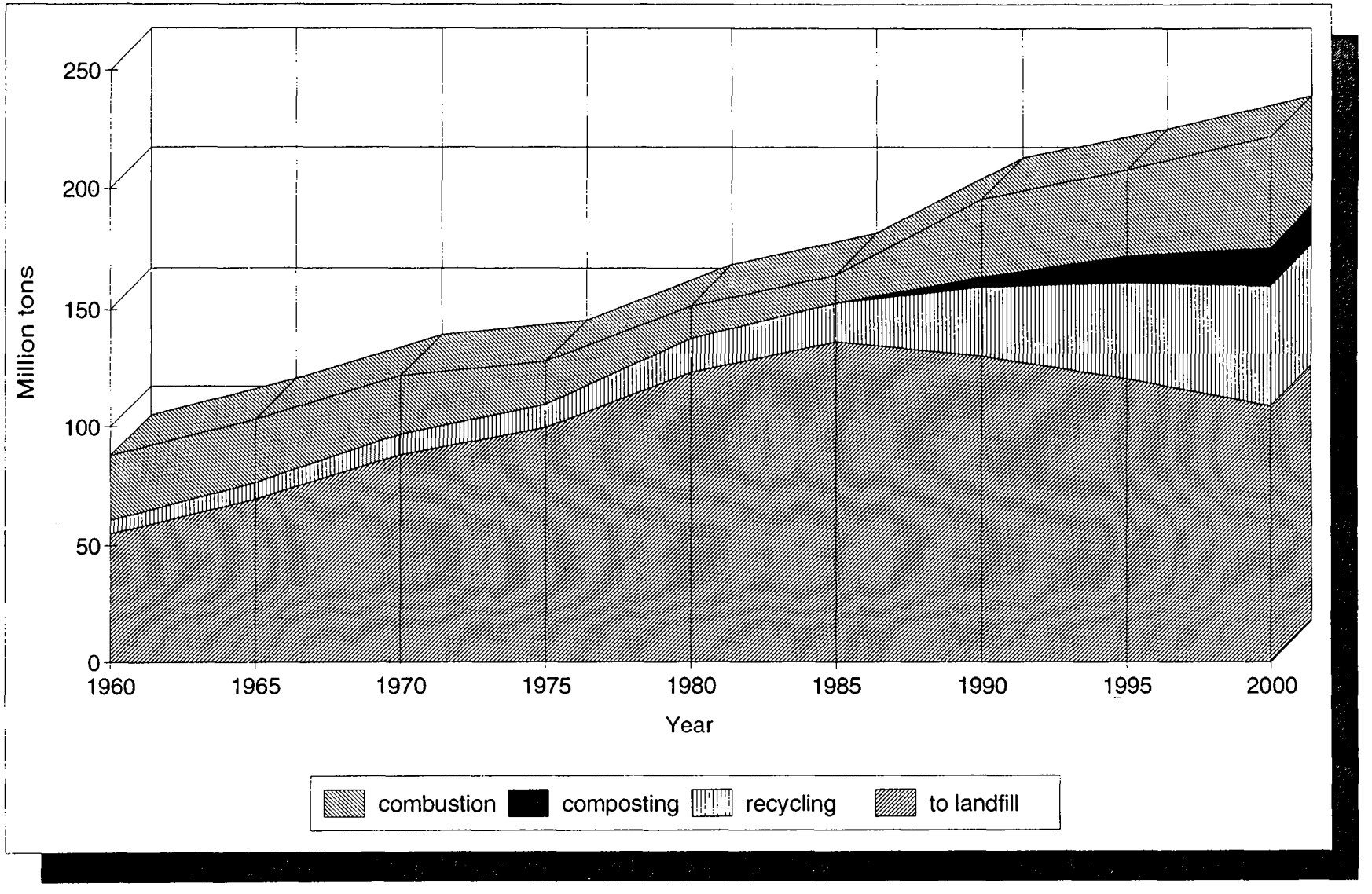

Figure 5. Trends in municipal solid waste management, 1960 to 2000.

TRENDS IN LANDFILL GAS RECOVERY AND UTILIZATION

Landfill gas recovery is a comparatively young industry: 75 percent of all landfill CH_4 utilization projects are less than seven years old (Governmental Advisory Associates, Inc., 1991). New technologies to utilize landfill gas are under development; these include application of fuel cells and the production of compressed gas vehicle fuel and possibly synthetic fuel. Fuel cells appear particularly promising and the EPA is funding further research to facilitate development of this technology (Sandelli, 1992). The conventional purification and combustion of CH_4, as it is applied in LFG operations, is a well known technology. It is unlikely that currently employed equipment can be further improved to notably enhance efficiency. On the other hand, it is plausible that significant improvements can be made upstream in the gas collection setup. The regulation of the flows from different wells or groups of wells is the most important factor in LFG recovery maximization. The lay-out and design of the collection system poses the biggest challenge in optimizing overall collection efficiency (Augenstein and Pacey, 1992).

In certain cases, LFG does not have to be converted to electrical energy before it can be sold. If there is an appropriate buyer, such as a nearby industrial plant, the gas can be sold directly and used to fuel the plant's boilers to offset the consumption of conventional fuels. This option requires the least capital investment since there is no need to tie into the local grid as is done by projects that generate electricity. In these situations, considerable gains in CH_4 recovery efficiency can be made by planning the project not only to maximize the profit, but also to maximize the amount of CH_4 recovered.

REGULATORY ISSUES AFFECTING LANDFILL GAS RECOVERY

Several different regulations affect LFG recovery. Some of these regulations are aimed at reducing hazardous emissions. Other regulations consist of federal or state incentives to promote the use of CH_4 from landfills as an energy source (Augenstein and Pacey, 1992).

A regulation has been proposed, directed at reducing emissions from certain new and existing landfills under the Clean Air Act Section 111(b) and 111(d) (Federal Register, May 30, 1991). The final rule will probably be promulgated by Summer 1994. This rule, requiring a gas collection system and add-on control device at affected landfills, is expected to result in the control of 500 to 700 sites, reducing 5 to 7 tg of CH_4 per year (Thorneloe, 1994b). Although energy conversion is not likely to be required in the final regulation, it would nevertheless be encouraged. Just as flares, LFG energy utilization projects reduce toxics, non-methane organic compounds and CH_4. Also, there are additional benefits associated with the utilization of this non-fossil fuel source, such as the potential offsets from coal-fired power plants.

To illustrate the significance of this potential energy source, comparisons were made using recent Department Of Energy statistics (DOE, 1993). For the year 2000, the potential energy of LFG for the sites affected by this rule is estimated to be equal to 2.7 percent of the annual U.S. coal consumption, comparable to 6.7×10^9 kWh. Electricity

generation with LFG projects, instead of coal-fired power plants and thereby offsetting fossil fuel consumption would result in a reduction of CO_2, SO_2, NOx, and other pollutants. Assuming recovery of all the gas that is available, there is potentially a savings of 4.2 million tons of CO_2, 1,300 tons of SO_2, and 12,000 tons of NOx (year 2000). Currently, data are insufficient to calculate the net reductions more accurately. EPA has research underway through its Office of Research and Development to develop a methodology for use by States in considering the offset in emissions associated with LFG utilization projects so that the overall environmental benefits of these projects can be considered in permitting applications.

Many other state and federal regulations exist regarding the control of LFG. State regulations proposed for California seem stricter than the federal regulation described above. California's draft guidelines propose that energy conversion projects must meet best available control technology (BACT) criteria (Augenstein and Pacey, 1992).

The U.S. Congress and several state legislative bodies have shown an intent to encourage and facilitate the use of energy from small-scale alternative sources such as LFG. This intent has resulted in various credits and incentives such as those in the Public Utility Regulatory Policies Act (PURPA). This Act provides the structure in which small-scale electricity producers can sell their power to utility grids. Under the act, utilities are required to buy the alternative electricity at "avoided cost," which is the sum of costs the utility would incur if it had to produce the energy itself.

Regulations also exist that hinder the development of LFG recovery projects. Under these regulations, the recovery systems are considered as sources of themselves. These regulations do not take into account the offset of pollutants from the landfill surface or energy savings associated with LPG recovery systems (Thorneloe, 1992b).

ESTIMATE OF FUTURE EMISSIONS

In summarizing and comparing the different trends in MSW generation, waste management, LFG utilization and pending legislation, two main trends can be identified: less MSW will be landfilled and gas recovery projects should become more feasible in the future. Due to legislative and economic pressure, there will be a tendency toward larger and fewer landfills. As small landfills begin to close in favor of larger landfills, there will be a shift from Size Class I into Size Class II and/or III. The regression lines developed for Size Class II and III have slopes that are less steep than the segment for Size Class I (Figure 2), so this would lead to a reduction of overall emissions.

The size classes and equations for the three segments of the Regression curve are:

I	$x < 1.128$
II	$1.128 \leq x < 4.082$
III	$x \geq 4.082$

where: x = welled waste, million metric tons or tg.

Innovation in gas collection systems and gas technology can be expected. This change, as well as the trend toward larger landfills will make gas recovery projects more feasible. Yet the main factor determining the feasibility of a LFG recovery project will remain the price for which the LFG, or power derived from it, can be sold. As a result of increased source reduction, recycling, composting and combustion the yearly amount of landfilled waste will continue to decrease (U.S. EPA, 1992), which will eventually lead to a reduction in annual CH_4 emissions rates from landfills. However, this is somewhat offset by increasing population.

However, the shifts towards larger landfills and the influence of changes in source reduction, recycling, composting and combustion may be overshadowed by the effect of the new landfill rule. This statement can be underwritten by the fact that, according to the Survey, 14 percent of all landfills had 1 million or more metric tons of waste in place. These sites produced 83 percent of all the landfill CH_4 emissions (1986). This illustrates that regulation of a small number of large landfills could possibly achieve a considerable reduction in CH_4 and other emissions from landfills.

SUMMARY AND CONCLUSIONS

The estimation of U.S. CH_4 emissions from landfilled waste is part of a bigger effort by AEERL, to obtain global greenhouse gas emissions data (Thorneloe, 1994a). Methane flow rates from landfills with LFG recovery systems can be used as surrogates for CH_4 generation and subsequently for CH_4 emissions. The AEERL in conjunction with the Solid Waste Association of North America collected data on 112 U.S. LFG recovery projects, 105 of which are included in the ORD Database (Appendix D).

The development of a regression model used for estimating CH_4 emissions, which relates LFG flow rates to waste in place data from the ORD Database, is described in this document. The model has three linear segments, each of which apply to a distinct landfill size class. A conversion factor was used to convert LFG flow rates to yearly CH_4 emissions. The conversion includes assumptions to account for the recovery efficiency of LFG projects and for the probable oxidation of CH_4 in the topsoil cover of the landfill.

In 1986, OSW conducted a survey, in which detailed information on 1,175 U.S. landfill facilities was compiled in a data base (OSW-Westat Database). This population was designed to be a stratified random sample of all U.S. landfills, therefore, its data can be extrapolated by means of scaling factors to obtain total waste in place for the U.S. This data base contains data which make it possible to estimate waste in place by two different methods, the RATE and the DIFF method. Preference is given to the DIFF method which is based on the difference between Total Design Capacity and Remaining Design Capacity of the landfill. According to the DIFF method, the total waste in U.S. landfills in 1986 was $4.7*10^{15}$ g ($5.2*10^9$ tons). The yearly disposal rate (1986–1992) was estimated to be 248 tg/yr.

Application of the regression model to the mass waste in place data calculated from the OSW-Westat Database yields national CH_4 emissions from landfills. This value is increased by the estimate of CH_4 emissions from industrial landfills and adjusted for CH_4 which is currently recovered or flared. An attempt has been made to update the emissions estimate for 1992. For 1986 CH_4 emissions from U.S. landfills were estimated at 11 tg/yr with lower and upper bound values of 7 and 15 tg/yr respectively. Methane emissions from U.S. landfills in 1992 were estimated to be 13 tg/yr with lower and upper bound values of 9 and 18 tg/yr.

This report details uncertainties which limit the quality of the above estimates. The main uncertainty arises from the quality of the waste in place data from the OSW-Westat Database. There are several indications that, since its publication, the data base has been subject to alterations: the scaling factors seem too high, waste in place data do not match up, and a density conversion has taken place. The original data base is not available and none of the alterations to the original data base have been documented; consequently, a quality assurance review of the data could not be performed. The possible uncertainties are inflated by the update from 1986 to 1992. Regarding the conversion factor, the efficiency of the gas recovery system appears to be the largest cause of uncertainty.

Due to regulatory and economic pressure, there will be a tendency toward larger and fewer landfills. Landfill gas recovery projects become more feasible as the landfill size increases, which should lead to a reduction in CH_4 emissions. As a result of source reduction, increased recycling, composting and incineration the yearly amount of landfilled waste is projected to continue to decrease; this will also lead to a reduction in annual CH_4 emissions from landfills. The influence of changes in waste management will likely be overshadowed by the effect of the anticipated new rule for controlling air emissions from landfills. This rule, requiring a gas collection system and add-on control device at affected landfills, is expected to result in the control of 500 to 700 sites, reducing CH_4 emissions by 5 to 7 tg per year.

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APPENDIX A: STATISTICAL METHODS

INTRODUCTION

The statistical methods employed to obtain estimates of total waste in place and national methane emissions are described in this appendix.

There are two independent sources of data from which the estimates are derived. The OSW-Westat Database is used to construct estimates of the total waste in place and the national landfill size distribution. The ORD Database is required, in conjunction with the OSW-Westat Database, to produce estimates of national methane emissions.

Landfill size and gas flow rate are denoted generically by x and y, respectively. Statistical estimates are reported for two methods of calculating waste in place x, the RATE and DIFF methods, as described in the text. The statistical theory is the same for both methods so that no distinction between the methods is made henceforth.

The value of x for the j^{th} landfill in the i^{th} stratum of the OSW-Westat Database is denoted x_{ij} , where $j = 1,...,n_i$, and $i = 1,2$; $i = 1$ denotes a small landfill and $i = 2$ denotes a large landfill. These designations were determined for each landfill in a preliminary classification of landfills reported in the Survey. Thus, n_i is the number of landfills surveyed in the i^{th} stratum. Strata 1 and 2 consist of small and large facilities respectively as determined by a preliminary classification of the landfills in the Survey.

Associated with each landfill in the OSW-Westat Database is a scaling factor, r_{ij}, which is simply the inverse of the stratum sampling fraction. The Survey reports that the target sampling fractions were 52 percent of the 'large' facilities and 13 percent of the 'small' facilities.

Thus the target sampling fractions suggest that the scaling factors for all landfills in the first stratum (small landfills) should be 1/0.13 = 7.69, and for the second stratum (large landfills), the scaling factors should be 1/0.52 = 1.92. The scaling factors reported in the OSW-Westat Database are for large and small landfills are 7.00 and 2.00, respectively (with a few values of 1 and 3 that must be erroneous). The source of the discrepancy between the recorded and anticipated scaling factors is unknown. Possibly the difference is due partly to a better-than-expected response rate, partly to rounding (for convenience perhaps), and perhaps in part to a recalculation of the number of 'eligible' landfills as described in the Survey.

The scaling factors used in the formulas that follow are those recorded in the OSW-Westat Database with the modification that the values of 1 and 3 were set equal to 2.00. Thus, r_{ij} = 7.00 for small landfills (*i.e.*, = 1) and r_{ij} = 2.00 for large landfills (*i.e.*, i =2).

The following quantities figure prominently in the formulas that follow:

$$N_i = \sum_{j=1}^{n_i} r_{ij}, \qquad i = 1,2;$$
$$N = N_1 + N_2; \tag{8}$$
$$n = n_1 + n_2.$$

Where n_1 and n_2 are the number of small and large landfills, respectively, in the Subtitle D Municipal Landfill Survey. With this notation, the total number of U.S. landfills (also an estimate, as described in the Subtitle D Municipal Landfill Survey) is N; similarly N_1 and N_2 are estimated stratum sizes.

The estimate of total waste in place is

$$\hat{X} = N_1\bar{x}_1 + N_2\bar{x}_2 , \tag{9}$$

where:

$$\bar{x}_{i\cdot} = \frac{1}{n_i}\sum_{j=1}^{n_i} x_{ij}, \qquad i = 1,2. \tag{10}$$

and: x_{ij} = the amount of waste in place in landfill j in stratum i
n_i = the target sampling fraction for stratum i.

This formula calculates total waste in place as the sum of the estimated waste at small and large landfills. Estimated waste in place at each landfill type is the product of the average waste in place of the surveyed landfills and the total number of landfills, by stratum.

The standard error of the estimate is calculated form the formula

$$s_{\hat{X}} = \sqrt{\left(\frac{N_1(N_1 - n_1)}{n_1}s^2{}_1 + \frac{N_2(N_2 - n_2)}{n_2}s^2{}_2\right)} \tag{11}$$

where s_i^2 is the sample variance of $x_{i1},\ldots,x_{in(i)}$, i = 1, 2. Applying these formulas to the data produces the results in Table 9.

TABLE 9. ESTIMATES OF TOTAL WASTE IN PLACE; STATISTICAL CONSIDERATIONS

	Method			
			Comparison	
	DIFF	RATE	DIFF	RATE
Landfills in Data Base	1,175			
Landfills with Complete Records	1,064	1,064	1,034	
Total Waste in Place (g) (standard error)	$4.86*10^{15}$ ($4.73*10^{14}$)	$6.61*10^{15}$ ($5.04*10^{14}$)	$4.72*10^{15}$ ($4.72*10^{14}$)	$6.36*10^{15}$ ($5.03*10^{14}$)
Estimated Landfills (N)	6,363	6,376	6,223	6,223

Notes:
- Standard error in parentheses. Approximate 95 percent confidence intervals are obtained by adding plus/minus two standard deviations to the estimate. Thus for the Comparison/DIFF method the estimate of total waste in place is 4,717 tg. The 95 percent confidence interval is (between 3,773 and 5,661 tg).
- Method denotes the various ways of accommodating incomplete records as described in the text.

Remarks:

1. The statistical theory is based on the assumption that N_1 and N_2 are known, not estimates as they are here. The effect of using estimated strata sizes is to increase the variability of the estimate. Consequently the standard errors reported in Table 9 are optimistic, *i.e.*, they undervalue the variability in the estimate of total waste in place.

2. The statistical theory also supposes that x is measured precisely, *i.e.*, there are no estimation errors or reporting errors, either systematic (bias) or random (noise) in the values of x_{ij}, *i.e.*, the estimates of waste in place. Random errors increase the variability of the estimate although the increased variability should, to a great extent, be reflected in the reported standard error. Systematic errors are more problematic. It is not possible to evaluate the effect of systematic errors with the available data.

3. The estimate of total waste in place, $\hat{X}$, may also be calculated as:

$$\hat{X} = \sum_{i=1}^{2} \sum_{j=1}^{n_i} r_{ij} x_{ij}. \qquad (12)$$

This representation makes clear the role of the scaling factors as weights in a weighted sum of the x_{ij} from the surveyed landfills.

ESTIMATING CH_4 USING REGRESSION RESULTS

The key variates in the ORD Database are y, the LFG flow rate, and x, the estimates of waste in place. Since these data are not stratified, values of y and x for an individual landfill are denoted with a single subscript, *e.g.*, y_j and x_j, $j = 1,...m$ where m is the number of landfills in the ORD Database. The ORD Database contains information on 112 landfills; however, seven landfills have missing values for either x or y and thus data from only 105 landfills are used in the statistical analyses.

Two methods were examined, Ratio estimation and Regression modeling. Ratio estimation is the simpler of the two; it generates a straight line running through the origin. There is some evidence that the Ratio method is not appropriate for the available U.S. data. The Regression modeling approach was adopted to overcome deficiencies with the Ratio method. It uses different emission factors for different landfill sizes. Consequently, it can only be used when country-specific information about landfill size is available, as is the case in the U.S.

Ratio Estimation

The ratio estimate of total landfill CH_4 emissions is

$$\hat{Y} = CF\hat{R}\hat{X} \tag{13}$$

where CF is the conversion factor described in Section 2; $\hat{X}$ is the estimate of total waste in place from (9), and

$$\hat{R} = \bar{y} / \bar{x} \tag{14}$$

which is simply the average LFG flow rate emitted per ton of landfilled waste. Therefore the Ratio method can be viewed as an emission factor.

$\bar{y}$ and $\bar{x}$ are calculated from

$$\bar{y} = \frac{1}{m}\sum_{j=1}^{m} y_j; \quad \textit{and} \quad \bar{x} = \frac{1}{m}\sum_{j=1}^{m} x_j. \tag{15,16}$$

The regression line resulting from application of the Ratio method is shown in Figure 6. The regression curve developed by the Regression method, as in Figure 2, is included for comparison.

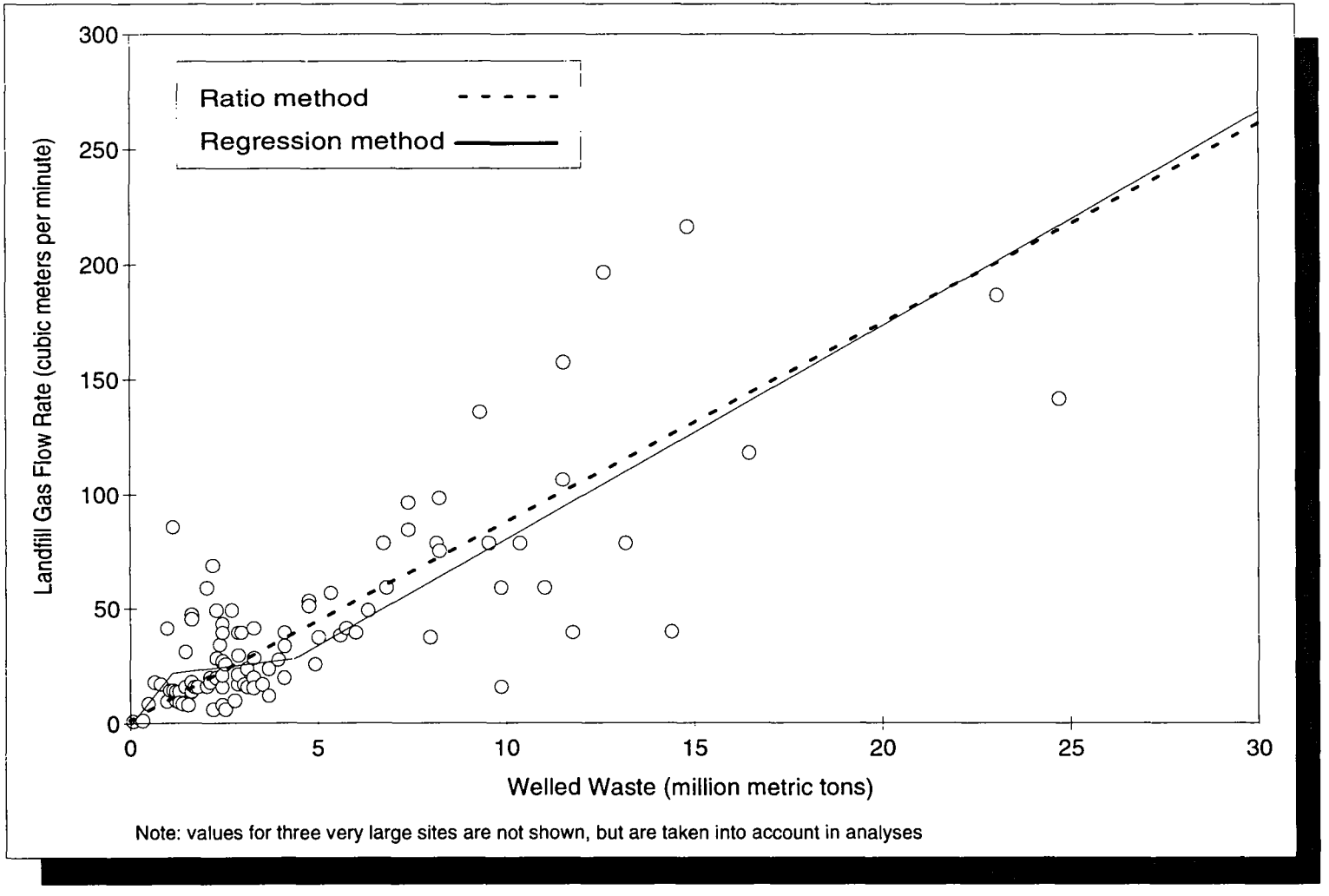

Figure 6. Landfill gas flow rates versus welled waste for 105 recovery sites and regression lines for Ratio and Regression methods.

The standard error of $\hat{Y}$ is computed according to the formula

$$s_{\hat{Y}} = CF\sqrt{\left(s_{\hat{R}}^2 s_{\hat{X}}^2 + \hat{X}^2 s_{\hat{R}}^2 + \hat{R}^2 s_{\hat{X}}^2\right)} \tag{17}$$

where $s_{\hat{X}}$ is the standard error of the estimate of total waste in place (11), and $s_{\hat{R}}$ is the standard error of $\hat{R}$ calculated from

$$s_{\hat{R}} = \sqrt{\left(\frac{1}{m\bar{x}^2}\sum_{j=1}^{m}(y_j - \hat{R}x_j)^2\right)} \tag{18}$$

Estimates of total yearly methane emissions obtained by ratio estimation are displayed in the Table 10.

TABLE 10. RATIO $\hat{R}$ AND RATIO ESTIMATES OF METHANE EMISSIONS FOR 1986.

	DIFF	RATE
Landfills in data base	1,175	
Landfills with complete records	1,034	
Total waste in place (g) (standard error)	$4.72*10^{15}$ $(4.72*10^{14})$	$6.36*10^{15}$ $(5.03*10^{14})$
Ratio $\hat{R} = \bar{y}/\bar{x}$ (m^3/g min) (standard error)	$8.76*10^{-12}$ $(0.79*10^{-12})$	
Gas flow rate (m^3/min) (standard error)	$4.13*10^4$ $(5.56*10^3)$	$5.59*10^4$ $(6.69*10^3)$
Conversion Factor (g min/yr m^3)	$213*10^6$	
Emissions estimate (g/yr) (standard error)	$8.81*10^{12}$ $(1.19*10^{12})$	$11.92*10^{12}$ $(1.42*10^{12})$
Emissions estimate for industrial landfills, (g/yr)	$0.55*10^{12}$	
Methane currently recovered or flared, (g/yr)	$1.1*10^{12}$	
Estimated total U.S. methane emissions, (g/yr)	$8.3*10^{12}$	$11.4*10^{12}$

Note: Standard errors are given in parentheses. The standard deviation in the emission factor, developed from the landfill gas flow data is 9 percent. Approximate 95 percent confidence intervals are obtained by adding plus/minus 18 percent.

Remarks:

1. Ratio estimation is justified statistically when either: (i) the landfills in the ORD Database can be regarded as a simple random sample of all landfills in the U.S.; or (ii) landfill gas flow rate is proportional to landfill waste in place, with the constant of proportionality independent of landfill size.

2. The landfills in the ORD Database were not randomly chosen from among all landfills in the U.S. and they are much larger than the averaged-sized landfill. Thus there is ample reason to suspect that (i) does not hold.

3. The second condition, (ii), seems reasonable; however, in light of the anecdotal evidence that CH_4 generation is greater in smaller landfills, (ii) is certainly suspect. Furthermore, the data tend to support the anecdotal evidence. Among the smaller landfills the majority of gas flow rates lie below the line, $\hat{Y} = \hat{R}\hat{X}$. However, the remaining smaller landfills vary substantially above the line, so much so that the average generation rate of the smaller landfills lies well above the line. This is evidence that the Ratio method *may* be inappropriate — the same behavior would also result from a tendency for operators of smaller landfills to under-report the amount of waste in place.

Regression Modeling

The ORD Database can be used to estimate a model for landfill gas flow rate as a function of landfill size. The model is used to 'predict' gas flow rate for each landfill in the data base, and then a national methane emission estimate is computed by summing the predicted gas flow rates over all landfills and applying the conversion factor. A weighted sum is employed as in the alternative definition of $\hat{X}$, equation (12).

Ratio estimation may be viewed as a special case of the method just described. If the appropriate gas flow rate model has the form

$$y = \beta x \tag{19}$$

that is, gas flow rate is proportional to the amount of waste in place, then an estimate of β can be obtained by averaging both sides of (19) over the values of y_j and x_j in the ORD Database. The resulting estimate would be

$$\hat{\beta} = \frac{\bar{y}}{\bar{x}} \tag{20}$$

which is just $\hat{R}$, the ratio estimate described previously in Equation (14). Next every landfill in the OSW-Westat Database is assigned the 'predicted' gas flow rate $\hat{y}_{ij} = \hat{\beta} x_{ij}$.

The weighted sum

$$CF \sum_{i=1}^{2} \sum_{j=1}^{n_i} r_{ij} \hat{y}_{ij}. \tag{21}$$

is an estimate of national emissions. Inspection of (21) shows that it is identical to $\hat{Y}$ defined in Equation (13).

The problems with ratio estimation discussed previously can be alleviated to a great extent by replacing the simple linear model (19) with a model that allows for differential gas flow rates depending on the size of the landfill. This can be accomplished with any number and variety of models. The particular model is not critical. The segmented model is sufficiently flexible to reflect the higher gas flow rates of the smaller landfills while providing an adequate fit to the larger landfills without unnecessary complexity.

The equation describing the piecewise linear model depends on five parameters, three representing the slopes of the linear segments, $\beta = (\beta_1, \beta_2, \beta_3)$ and two determining the 'hinge' points, $\theta = (\theta_1, \theta_2)$. These parameters were estimated by weighted least squares with weights proportional to the reciprocal of landfill sizes. In other words, if $\mu(x, \beta, \theta)$ denotes the function defining the piecewise linear model, then $\hat{\beta}$ and $\hat{\theta}$ minimize

$$\sum_{j=1}^{m} \frac{1}{x_j} \{ y_j - \mu(x_j, \beta, \theta) \}^2 \tag{22}$$

over all values of β *and* θ. This is analogous to the mathematical criterion for estimating β (equivalently R) when the simple linear model (19) is employed.

Given the similarity of the estimated piecewise linear model to the simple model $y = \hat{R}x$, it is natural to question whether the difference is significant, either practically or statistically. Table 11 displays estimates of national methane emissions derived from the estimated piecewise linear model.

TABLE 11. REGRESSION ESTIMATES OF METHANE EMISSIONS, STATISTICAL CONSIDERATIONS

	Method			
	DIFF	RATE	Comparison	
			DIFF	RATE
Landfills in Data Base	1,175			
Landfills with Complete Records	1,064	1,064	1,034	
Total Waste in Place (g) (standard error)	$4.86*10^{15}$ $(4.73*10^{14})$	$6.61*10^{15}$ $(5.04*10^{14})$	$4.72*10^{15}$ $(4.72*10^{14})$	$6.36*10^{15}$ $(5.03*10^{14})$
Gas Flow Rate (m^3/min) (standard error)	$5.39*10^{4}$ $(7.00*10^{3})$	$6.83*10^{4}$ $(8.20*10^{3})$	$5.25*10^{4}$ $(6.83*10^{3})$	$6.66*10^{4}$ $(7.91*10^{3})$
Conversion Factor (g min/yr m^3)	$213*10^{6}$			
Emissions Estimate (g/yr) (standard error)	$11.49*10^{12}$ $(1.50*10^{12})$	$14.95*10^{12}$ $(1.75*10^{12})$	$11.21*10^{12}$ $(1.46*10^{12})$	$14.21*10^{12}$ $(1.69*10^{12})$

Note: Standard errors are given in parentheses. Approximate 95 percent confidence intervals are obtained by adding plus/minus two standard deviations to the estimate.

Remarks:

1. The estimates in Table 11 are noticeably greater than those derived from the ratio method described previously. The reason is that the vast majority of landfills are small and even the slightest difference between models for small landfills will induce a large difference in national estimates.

2. The decision to use the piecewise linear model was based in part on the anecdotal evidence of higher generation rates at smaller landfills. However, in principle it is also possible to make a purely data-based comparison of the simple linear and piecewise linear models, that is, to perform a statistical test comparing the fit of the models. There is a problem in that the statistical theory for comparing piecewise linear models is generally less well understood than the theory for comparing linear models. Nevertheless the results of the test gives some guidance as to the appropriate choice of model. The normal-theory likelihood ratio test for comparing the linear and piecewise linear models indicates that the latter model fits the data significantly better (p-value = 0.02) than the simple linear model. However, this result should be interpreted cautiously, given the approximate nature of the statistical theory.

3. Calculating standard errors for the emission estimates derived from the more complicated model is difficult. The standard errors reported in Table 11 were

calculated as 13 and 12 percent of the emissions estimates for the difference and rate methods respectively. These percentages were derived from the standard errors and emissions estimates from the ratio estimates. The standard errors derived in this fashion should be reasonably accurate, although they are probably biased low.

4. The combination of sparseness of the data for large values of x and the heteroscedastic variation of gas flow rates causes problems when estimating the parameters of the piecewise linear model. The determination of the 'hinge' points is particularly sensitive to these characteristics of the data. Problems were avoided by minimizing (22) subject to an upper bound on the largest 'hinge' point.

5. The flexibility of the piecewise linear model also makes the estimated model more sensitive to outlying or extreme data points, and there are certain landfills in the ORD Database whose exclusion from the data have a pronounced effect on the estimated model and on the final emissions estimates. When suspected outliers are removed from the data the emissions estimates so calculated are generally in closer agreement with the ratio estimates. Since it was not possible to confirm that any suspected outliers were in fact erroneous, only the results for the complete data set are presented.

APPENDIX B: COMPARISON OF ESTIMATES OF METHANE EMISSIONS FROM U.S. LANDFILLS

TABLE 12. COMPARISON OF ESTIMATES OF METHANE EMISSIONS FOR U.S. LANDFILLS

SOURCE	EMISSIONS ESTIMATE (tg/yr)			REMARKS
	Lower bound	Mid-point	Upper bound	
Bingemer & Crutzen[1] (1987 estimate)	11	16	21	Pro-rated from global estimates. Uses a mass balance approach.
IPCC/OECD Method[2] (1990 estimate).		20		Uses same waste data as this study (for comparison).
Augenstein[3] (1990 estimate)	3	6	8	Uses a gas generation model, based on decomposition kinetics.
OAP, Global Change Division[4] (1990 estimate)	8	10	12	Uses a model similar to this study with lower waste in place estimates.
This study (1986)	7	11	15	
(1990)	8	13	17	
(1992)	9	13	18	

[1] Bingemer, H.G. and P.J. Crutzen. 1987. The Production of Methane from Solid Wastes. Journal of Geophysical Research, Vol. 92, No. D2.

[2] OECD (Organization for Economic Cooperation and Development). 1991. Estimation of Greenhouse Gas Emissions and Sinks. Final Report from OECD Experts Meeting, 18-21 February 1991, Paris, France. Prepared for Intergovernmental Panel on Climate Change. OECD, Paris, France.

[3] Augenstein, D.C. 1990. Greenhouse Effect Contributions of United States Landfill Methane. GRCDA 13th Annual Landfill Gas Symposium, Lincolnshire, Il.

[4] U.S. EPA, Office of Air and Radiation (OAR), Global Change Division. 1993. Anthropogenic Methane Emissions in the United States: Estimates for 1990. Report to Congress.

APPENDIX C: CALCULATIONS FOR THE DENSITY OF METHANE

Ideal gas law: $PV/nRT = \text{constant}$,

where

$R = 8314.41$ Joule per Kelvin per kilomole [J/(K.kmole)]

for

P = pressure in Pascal
V = volume in m^3
n = number of kmoles of gas
T = temperature in K.

Therefore, $V = nRT/P = 22.4138\ m^3$ for:

with

$n = 1$ kmole
$T = 273.15$ K (which = 0°C)
$P = 101{,}325$ Pa (which = 1 atm)

The molecular weight of 1 kmole of CH_4 = 16,040 g

So the density $\rho = 16{,}040/22.4138 = 715.631\ g/m^3$

To convert to a standard with T = 60°F = 288.71 K (and all other parameters unchanged), multiply by 273.15/288.71 which gives $\rho = 677.062\ g/m^3$.

APPENDIX D: FIELD DATA

TABLE 13. WASTE AND LFG FLOW RATE DATA FROM THE ORD DATABASE

Landfill Identification Code	Welled Waste (Millions of U.S. Tons)	Welled Waste (tg)	GAS (MCFD)	Gas Flow Rate (CFM[1])	Gas Flow Rate (CMM[2])
323	0.1	0.0	0.04	28	0.7
331	0.4	0.3	0.05	35	0.9
19	0.6	0.5	0.42	292	8.2
261	0.8	0.7	0.90	625	17.7
334	1.0	0.9	0.86	597	16.9
272	1.2	1.0	2.10	1458	41.3
298	1.2	1.0	0.48	333	9.4
320	1.3	1.1	0.72	500	14.1
208	1.4	1.2	4.36	3028	85.7
318	1.4	1.2	0.72	500	14.1
213	1.5	1.3	0.50	347	9.8
304	1.5	1.3	0.70	486	13.7
330	1.6	1.4	0.45	313	8.8
210	1.6	1.4	0.50	347	9.8
290	1.6	1.4	0.70	486	13.7
308	1.7	1.5	0.42	292	8.2
9	1.8	1.6	1.58	1097	31.0
232	1.8	1.6	0.80	556	15.7
244	1.9	1.7	0.40	278	7.8
284	2.0	1.8	0.90	625	17.7
255	2.0	1.8	2.40	1667	47.2
802	2.0	1.8	0.69	479	13.5
15	2.0	1.8	2.30	1597	45.2
337	2.0	1.8	0.70	486	13.7
360	2.1	1.9	0.80	556	15.7
278	2.2	1.9	0.80	556	15.7
302	2.5	2.2	3.00	2083	59.0
228	2.5	2.2	0.81	563	15.9
328	2.6	2.3	0.90	625	17.7
231	2.6	2.3	1.00	694	19.6
227	2.7	2.4	3.50	2431	68.8
263	2.7	2.4	0.30	208	5.9
342	2.8	2.5	1.00	694	19.6
313	2.8	2.5	2.50	1736	49.1
315	2.8	2.5	1.44	1000	28.3

(continued)

TABLE 13. WASTE AND LFG FLOW RATE DATA FROM THE ORD DATABASE (Continued)

Landfill Identification Code	Welled Waste (Millions of U.S. Tons)	Welled Waste (tg)	GAS (MCFD)	Gas Flow Rate (CFM[1])	Gas Flow Rate (CMM[2])
317	2.9	2.6	1.73	1201	34.0
8	3.0	2.7	1.05	729	20.6
332	3.0	2.7	2.00	1389	39.3
221	3.0	2.7	2.20	1528	43.2
355	3.0	2.7	1.37	951	26.9
316	3.0	2.7	0.40	278	7.8
247	3.0	2.7	0.80	556	15.7
23	3.1	2.8	0.30	208	5.9
299	3.1	2.8	1.30	903	25.5
6	3.1	2.8	1.30	903	25.5
305	3.3	2.9	2.50	1736	49.1
307	3.4	3.0	0.50	347	9.8
11	3.5	3.1	1.08	750	21.2
257	3.5	3.1	2.00	1389	39.3
7	3.5	3.1	1.50	1042	29.5
239	3.5	3.1	0.86	597	16.9
240	3.5	3.1	1.08	750	21.2
258	3.6	3.2	2.00	1389	39.3
336	3.6	3.2	2.00	1389	39.3
218	3.7	3.3	0.85	590	16.7
295	3.8	3.4	0.80	556	15.7
238	3.8	3.4	1.20	833	23.6
209	4.0	3.6	1.00	694	19.6
16	4.0	3.6	2.10	1458	41.3
312	4.0	3.6	0.78	542	15.3
220	4.0	3.6	1.44	1000	28.3
285	4.0	3.6	1.00	694	19.6
279	4.3	3.9	0.85	590	16.7
233	4.5	4.0	1.20	833	23.6
241	4.5	4.0	0.60	417	11.8
286	4.8	4.3	1.40	972	27.5
274	5.0	4.5	1.00	694	19.6
219	5.0	4.5	1.70	1181	33.4
354	5.0	4.5	2.00	1389	39.3
10	5.8	5.2	2.71	1882	53.2
326	5.8	5.2	2.60	1806	51.1
300	6.0	5.4	1.30	903	25.5

(continued)

TABLE 13. WASTE AND LFG FLOW RATE DATA FROM THE ORD DATABASE (Continued)

Landfill Identification Code	Welled Waste (Millions of U.S. Tons)	Welled Waste (tg)	GAS (MCFD)	Gas Flow Rate (CFM[1])	Gas Flow Rate (CMM[2])
234	6.1	5.5	1.90	1319	37.3
314	6.5	5.8	2.88	2000	56.6
321	6.8	6.1	1.94	1347	38.1
294	7.0	6.3	2.10	1458	41.3
24	7.3	6.6	2.00	1389	39.3
2	7.7	6.9	2.50	1736	49.1
1	8.2	7.4	4.00	2778	78.6
352	8.3	7.5	3.00	2083	59.0
229	9.0	8.1	4.30	2986	84.5
246	9.0	8.1	4.90	3403	96.3
237	9.7	8.7	1.90	1319	37.3
335	9.9	8.9	4.00	2778	78.6
216	10.0	9.0	5.00	3472	98.3
291	10.0	9.0	3.82	2653	75.1
18	11.3	10.2	6.90	4792	135.7
3	11.6	10.5	4.00	2778	78.6
289	12.0	10.8	0.80	556	15.7
207	12.0	10.8	3.00	2083	59.0
25	12.6	11.4	4.00	2778	78.6
245	13.4	12.1	3.00	2083	59.0
327	14.0	12.6	5.40	3750	106.2
343	14.0	12.6	8.00	5556	157.3
214	14.3	12.9	2.00	1389	39.3
4	15.3	13.8	10.00	6944	196.6
217	16.0	14.5	4.00	2778	78.6
347	17.5	15.8	2.02	1403	39.7
230	18.0	16.3	11.00	7639	216.3
275	20.0	18.1	6.00	4167	118.0
249	25.0	22.6	23.00	15972	452.3
14	28.0	25.3	9.50	6597	186.8
303	30.0	27.2	7.20	5000	141.6
251	60.0	54.4	37.50	26042	737.5
226	66.0	59.8	14.40	10000	283.2

[1] CFM = cubic feet per minute.
[2] CMM = cubic meter per minute.

APPENDIX D: FIELD DATA (continued)

TABLE 14. SUMMARY OF LANDFILL DATA FROM PEER ET AL. (1992)*

Landfill Number	Refuse Mass (10^6 Mg)	Average Refuse Age (yrs)	Average Methane Recovery Rate (m^3/min)	Average Methane Recovery Rate Per Unit Mass (m^3/min/10^6 Mg)	Average Temp. (° C)	Average Dewpoint (° C)	Annual Rainfall (cm)
1	6.35	8.0	55.3	8.71	7.50	3.33	73.05
2	6.12	10.0	18.0	2.94	9.28	3.89	90.47
3	7.35	10.0	40.0	5.45	12.39	6.11	105.21
4	13.79	9.5	98.4	7.14	23.94	18.33	155.91
5	10.89	15.0	24.8	2.28	17.11	10.00	43.23
6	2.40	7.0	16.7	6.97	16.17	8.33	45.44
7	2.95	10.0	9.7	3.28	15.00	8.89	106.07
8	2.72	10.0	11.7	4.32	16.22	10.00	123.47
9	1.63	7.0	7.7	4.71	18.11	12.22	134.16
10	5.26	12.0	29.3	5.57	16.78	10.56	138.48
11	1.81	10.0	11.3	6.22	18.17	11.11	113.94
12	2.78	8.5	8.0	2.87	12.17	6.67	122.94
13	0.96	7.0	10.4	10.87	11.06	6.67	104.32
16	3.38	5.5	16.0	4.74	9.50	2.78	102.01
17	5.17	10.0	13.8	2.67	8.22	1.67	120.90
20	9.71	11.0	35.0	3.61	9.22	3.89	78.66
21	2.60	13.0	27.4	10.52	9.22	3.89	78.66
22	3.97	12.0	33.2	8.35	7.06	1.11	66.95
23	2.87	10.7	2.2	0.78	10.17	-2.22	38.89
24	6.21	5.6	17.7	2.85	18.89	11.11	74.80
25	10.65	12.0	20.2	1.90	9.56	3.89	84.68

* Peer, R.L., D.L. Epperson, D.L. Campbell, and P. von Brook. 1992. Development of an Empirical Model of Methane Emissions from Landfills. U.S. EPA, Air and Energy Engineering Research Laboratory, Research Triangle Park, NC. EPA-600/R-92-037 (NTIS PB92-152875).