United States
Environmental Protection
Agency

Office of Air Quality
Planning and Standards
Research Triangle Park NC 27711

EPA-450/3-90-011a
March 1991

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Air Emissions from Municipal Solid Waste Landfills - Background Information for Proposed Standards and Guidelines

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Emission Standards Division

U.S. ENVIRONMENTAL PROTECTION AGENCY
Office of Air and Radiation
Office of Air Quality Planning and Standards
Research Triangle Park, North Carolina 27711
March 1991

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Publication No. EPA-450/3-90-011a

ENVIRONMENTAL PROTECTION AGENCY

Background Information Document
for Air Emissions from Municipal Solid Waste Landfills

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3/14/91
(Date)

1. The standards of performance and emission guidelines limit emissions from new and existing municipal solid waste landfills to 150 Mg/year of non-methane organic compounds (NMOC's). Section 111 of the Clean Air Act (42 U.S.C. 7411), as amended, directs the Administrator to establish standards of performance and emission guidelines for any category of source of air pollution that "... causes or contribute significantly to air pollution which may reasonably by anticipated to endanger public health or welfare."

2. Copies of this document have been sent to the following Federal Departments: Office of Management and Budget, Commerce, Interior, and Energy; the National Science Foundation; and the Council on Environmental Quality. Copies have also been sent to members of the State and Territorial Air Pollution Program Administrators; the Association of Local Air Pollution Control Officials; EPA Regional Administrators; and other interested parties.

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1. INTRODUCTION

This document supports legislative action taken by the U.S. Environmental Protection Agency (EPA) under Sections 111(b) and 111(d) of the Clean Air Act (CAA) (42 U.S.C. 1857 et seq), as amended, to control air emissions from municipal solid waste landfills (hereafter referred to as municipal landfills) as defined in Subtitle D of the Resource Conservation and Recovery Act. Section 111 directs the Administrator to establish standards of performance for any category of new stationary source which"... causes, or contributes significantly to air pollution which may reasonably be anticipated to endanger public health or welfare." Municipal landfill air emissions are being regulated because of the adverse health and welfare impacts caused by the following characteristics of landfill gas: (1) presence of volatile organic compounds; (2) presence of toxic and potentially hazardous compounds; (3) explosion potential; and (4) odor nuisance.

Standards of performance for stationary sources are required to reflect"... the degree of emission reduction achievable which (taking into account the cost of achieving such an emission reduction, and any nonair quality health and environmental impacts and energy requirements) the Administrator determines has been adequately demonstrated for that category of sources." The standards developed under Section 111(b) apply only to new stationary sources that have been constructed or modified after regulations are proposed by publication in the Federal Register.

Under Section 111(d), EPA has established procedures whereby States submit plans to control existing sources of "designated pollutants." Designated pollutants are those which are not included on a list published under Section 108(a) or 112(b)(1)(a), but to which a standard of performance applies under Section 111(b). Section 111(d) requires emission standards to be adopted by the States and submitted to EPA for approval. The standards would limit emissions of designated pollutants from existing facilities, which would be subject to the standards of performance for new stationary sources if they were new sources.

Subpart B of 40 CFR 60 contains the procedures under which States submit these plans to control existing sources of designated pollutants. Subpart B requires the States to develop plans for the control of designated pollutants within Federal guidelines. As indicated in Subpart B, EPA will publish guidelines for development of State emissions standards for a designated pollutant. These guidelines will apply to designated facilities which emit those designated pollutants and will include useful information for States, such as discussion of the pollutant's effects, description of control techniques and their effectiveness, costs, and potential impacts. Finally, as guidance for the States, recommended emission guidelines and times for compliance are identified.

The chapters of this document present the technical information on which the legislative actions under Sections 111(b) and 111(d) are based. They also present the necessary information discussed above for States to consider in establishing standards for existing municipal landfills.

Chapter 2 provides background information on the health and welfare impacts of municipal landfill air emissions. This includes the cancer and noncancer health effects of components in landfill gas; documented cases where explosions and fires have occurred; and studies listing identified problems with odors emanating from landfills.

Chapter 3 provides an overview of municipal landfill characteristics and discusses their emission potential. It describes the mechanisms by which emissions occur; quantifies baseline VOC emissions from new and existing landfills; quantifies the typical concentration of hazardous compounds; and details the ways in which explosion hazards and odor nuisance problems can occur.

Chapter 4 presents the techniques for controlling municipal landfill air emissions. This includes details on achieving the efficient collection of landfill gas; applicability and efficiency of available control systems; and potential byproduct emissions.

Chapter 5 presents the alternatives for regulating new and existing landfills. This section includes a discussion of the derivation of the regulatory alternatives and the corresponding impacts on existing and new municipal solid waste landfills.

Chapter 6 quantifies the health, welfare, environmental (air and water pollution) and energy impacts for each regulatory alternative.

Chapter 7 presents the estimated costs of controlling municipal landfill air emissions. This includes the design features of the collection and control system as well as the basis for capital and annual operating costs. The approach for estimating nationwide cost impacts is also discussed

Chapter 8 presents the economic impacts ... [complete after the chapter is finished].

Chapter 9 provides a description of the emission guidelines and compliance schedule for States to follow.

EPA-450/3-90-011a

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Emission Standards Division

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Research Triangle Park, North Carolina 27711
July 1990

2. HEALTH AND WELFARE EFFECTS OF AIR EMISSIONS FROM MUNICIPAL SOLID WASTE LANDFILLS

2.1 INTRODUCTION

This chapter presents a summary of the potential adverse health and welfare effects of air emissions from municipal solid waste (MSW) landfills. The five major effects of MSW landfill air emissions are (1) human health and vegetation effects caused by ozone formed from nonmethane organic compound (NMOC) emissions, (2) carcinogenicity and other possible noncancer health effects associated with specific MSW landfill emission constituents, (3) global warming effects from methane emissions, (4) explosion hazards, and (5) odor nuisance. In addition, soils and vegetation on or near the landfills are adversely affected by MSW landfill emissions migrating through the soil. The above effects are briefly summarized below and in Table 2-1.

A variety of different NMOCs have been detected in air emissions from MSW landfills. In the atmosphere, NMOCs can contribute to formation of ozone through a series of photochemical reactions. The ozone formed through these reactions can exert adverse effects on human health and on vegetation. The effects ozone exerts on both human health and vegetation are discussed in greater detail in Section 2.2.

There are potential acute and chronic health hazards associated with several chemical species in MSW landfill emissions. The potential cancer risks associated with exposure to MSW landfill emissions have been considered by EPA (see Section 2.3). There are also other chronic noncancer health effects associated with some of the individual chemicals found in MSW landfill air emissions. Qualitative descriptions of both the cancer and noncancer health effects are also included in Section 2.3.

The landfill gas that is generated from the decomposition of municipal solid waste in a landfill consists of approximately 50 percent methane and 50 percent carbon dioxide, and less than 1 percent NMOCs. The methane emissions are of concern for two reasons: 1) methane, one of the "greenhouse gases", contributes to the phenomenon of global warming

TABLE 2-1. SUMMARY OF THE HEALTH AND WELFARE EFFECTS ASSOCIATED WITH MSW LANDFILL EMISSIONS AND COMPONENTS

Component	Health and welfare effects
Ozone	Alterations in pulmonary function, aggravation of pre-existing respiratory disease, damage to lung structure; foliar injury, such as stippling or flecking, reduced growth, decreased yield
Toxics	Leukemia, aplastic anemia, multiple myeloma, cytogenic changes, damage to liver, lung, kidney, central nervous system, possible embryotoxicity, brain, liver and lung cancer, possible teratogenicity
Methane	Death, burns, dismemberment due to explosions and fires; property damage; contribution to phenomenon of global warming; MSW landfill emissions migrating through the soil on or near the landfill inhibits revegetation, causing deep root death
Odor	Odor nuisance, leading to annoyance, irritability, tension, reduction in outdoor activities, reduction in property values, decreased commercial investment leading to decreased sales, tax revenue

(Section 2.4); and 2) the accumulation of methane gas in structures both within and beyond the landfill boundary has resulted in explosions, fires, and subsequent loss of property (Section 2.5).

Pollutants that exert effects on human welfare are pollutants that affect the quality of life, cause damage to structures, or result in a loss of vegetation. The welfare effects of concern associated with MSW landfill air emissions include, in addition to destruction of property by explosions, emanation of odors and effects on soil and vegetation. Although odor perception is extremely variable and subjective, sociological studies have shown extreme annoyance and emotional disturbances in individuals residing in areas where objectionable odors are present. Property values may decrease and economic disadvantages may result in communities in or near a source of perceived malodorous emissions such as those from MSW landfills. Section 2.6 discusses odor generation by MSW landfills and some of the studies and surveys that have been done about the problem of odor nuisance. Also, revegetation of uncontrolled landfills after closure is often unsuccessful because the landfill gases affect plant root structure. This effect is discussed in Section 2.7.

2.2 EFFECTS ON HUMAN HEALTH AND VEGETATION CAUSED BY AMBIENT OZONE FORMED FROM NONMETHANE ORGANIC EMISSIONS

2.2.1 Health Effects Associated with Exposure to Ozone

Ozone and other oxidants found in ambient air are formed as the result of atmospheric physical and chemical processes involving two classes of precursor pollutants, NMOCs and nitrogen oxides (NO_x). NMOCs are constituents of the air emissions from MSW landfills. Therefore, emissions of NMOCs from landfills also contribute to ozone formation. The effects of ozone on human health are well documented. There are several different mechanisms through which ozone can exert adverse effects on human health. Ozone can penetrate into different regions of the respiratory tract and be absorbed through the respiratory system. Indirect effects of ozone are those such as adverse effects on the pulmonary system resulting from chemical interactions of ozone as it progresses through the system. Finally there may be adverse effects on other body organs and tissues caused indirectly by reactions of ozone in the lungs.[1]

Specific adverse human health effects associated with exposure to ozone include:[2]

- changes in pulmonary function;
- symptomatic effects;
- aggravation of pre-existing respiratory disease;
- damage to the lung structure;
- increases in susceptibility to respiratory infections; and
- adverse effects on blood enzymes, central nervous system, liver and endocrine system.

Pulmonary function decreases have been reported in healthy adult subjects after one to three hours of exposure to ozone. Subjects at rest (not exercising) have shown decreases in lung function at concentrations of about 0.5 ppm ozone.[3] Persons that are heavily exercising have experienced decreases in lung function at about 0.1 ppm ozone.[4]

Symptomatic effects, such as cough, shortness of breath, general trouble in breathing and pain when breathing have been reported in controlled human exposure studies. These effects reportedly occurred when exposure levels exceeded an ozone concentration of 0.12 ppm.[5]

There is some indication from a group of epidemiological studies that persons with existing respiratory diseases may experience aggravation of their conditions when exposed to ozone. Definitive data correlating increased rates of asthma attacks to ozone exposure do not exist, however.[6]

Another possible effect of ozone exposure is damage to the lung structure. Laboratory studies of rats and monkeys have shown inflammation and damage to lung cells following exposure to ozone. Studies on rats, mice, and rabbits have shown increased susceptibility of the animals to bacterial respiratory infections following ozone exposure.[7] Considering the differences in human and animal physiology and immune defenses, it is still reasonable to hypothesize that humans exposed to ozone could experience increased susceptibility to respiratory infections.[8]

Finally, some animal studies have indicated that exposure to ozone exerts adverse effects on the cardiovascular, liver and endocrine systems.

Definitive data on humans to substantiate these occurrences are not available. However, the body of evidence from the animal studies suggests that ozone can cause effects in tissues and organs other than the lung.[9]

2.2.2 Adverse Effects of Ozone on Vegetation

Foliar injury on vegetation is one of the earliest and most obvious manifestations of ozone impacts. The specific effects can range from reduced plant growth and decreased yield, to changes in crop quality and alterations in susceptibility to abiotic and biotic stresses. The plant foliage is the primary site of ozone effects, although significant secondary effects, including reduced growth and yield, can occur. Ozone injury to foliage is identified as a stippling or flecking. Such injury has occurred experimentally in various plant species after exposure to 60 ug/m^3 (0.03 ppm) ozone for 8 hours.[10] Studies with tobacco and other crops confirmed that ozone injures vegetation at sites near urban centers.[11] It is now recognized that vegetation at rural sites may be injured by ozone that has been transported long distances from urban centers.[12] Studies of the effect of ozone on plant growth and crop yield indicate occurrences of detrimental effects. For example, field studies in the San Bernadino Forest during the last 30 years show that ambient ozone has reduced the height growth of Ponderosa pine by 25 percent and has reduced the total volume of wood produced by 84 percent.[13]

2.3 CANCER AND NONCANCER HEALTH EFFECTS

The adverse human health effects associated with MSW landfill emissions have not been directly determined by human or animal studies. In the absence of such data, EPA has evaluated some of the individual chemical constituents of MSW landfill emissions. Over 100 chemical constituents have been detected in MSW landfill emissions, as shown in Table C-1 of Appendix C. Exposure to the several of the landfill constituents has been associated with cancer and noncancer health effects. Both cancer and noncancer health effects have not been quantified in a national study, due to limitations in the emissions data. However, these adverse effects can be discussed qualitatively. Adverse effects on target organ systems such as the kidney, liver, pulmonary, and central nervous systems have been

associated with various components of air emissions from MSW landfills. A detailed summary of the health effects is given in in Section 2.3.1.2.

2.3.1 Hazard Identification

Hazard identification is a qualitative step for determining whether or not exposure to a given substance is associated with any adverse health effect. Because epidemiological and animal studies of the health effects of MSW landfill emissions, which are mixtures of many chemicals, have not been found, the hazard identification process was based on a review of the health data of the MSW components. This review focused on nine carcinogenic constituents known to be present in MSW landfill air emissions (benzene, carbon tetrachloride, chloroform, ethylene dichloride, methylene chloride, perchloroethylene, trichloroethylene, vinyl chloride, and vinylidene chloride--Table 2-2). There were other carcinogenic compounds emitted, but these nine pollutants have been repeatedly measured in the air emissions from various MSW landfills.

One of the initial steps that EPA takes in addressing the potential for health effects is to consider the quality of the available data for each MSW landfill gas constituent. The EPA has developed a classification scheme for characterizing the weight-of-evidence for human carcinogenicity. Evidence of possible carcinogenicity in humans comes primarily from two sources: long-term animals tests and epidemiologic investigations. Results from these studies are supplemented with available information from other relevant toxicologic studies. The question of how likely an agent is to be a human carcinogen is answered in the framework of a weight-of-evidence judgment. Judgments about the weight of evidence involve considerations of the quality and adequacy of the data and the kinds and consistency of responses induced by a suspect carcinogen. There are three major steps to characterizing the weight-of-evidence for carcinogenicity in humans: (1) characterization of the evidence from human studies and from animal studies individually, (2) combination of the characterizations of these two types of data into an indication of the overall weight-of-evidence for human carcinogenicity, and (3) evaluation of all supporting information to determine if the overall weight-of-evidence should be modified.

TABLE 2-2. SUMMARY OF THE HEALTH EFFECTS ASSOCIATED WITH TOXIC MSW LANDFILL EMISSIONS COMPONENTS

Component	Health and welfare effects
Benzene	Leukemia, aplastic anemia, multiple myeloma, cytogenic changes--human carcinogen
Carbon tetrachloride	Damage to liver, lung, kidney, central nervous system. Possible embryotoxicity--probable human carcinogen
Chloroform	Damage to liver, kidney, central nervous system--probable human carcinogen
Ethylene dichloride	Damage to central nervous system--probable human carcinogen
Methylene chloride	Probable human carcinogen
Perchloroethylene	Probable human carcinogen
Trichloroethylene	Probable human carcinogen
Vinyl chloride	Central nervous system effects; brain, liver and lung cancer; possible teratogen--human carcinogen
Vinylidene chloride	Damage to liver, kidney--possible human carcinogen

The EPA has developed a system for stratifying the weight-of-evidence. This classification is not meant to be applied rigidly or mechanically. At various points in the above discussion, there is the need for an overall, balanced judgment of the totality of the available evidence. Particularly for well-studied substances, the scientific data base will have a complexity that cannot be captured by any classification scheme. The EPA's weight-of-evidence system is summarized in Table 2-3 and the specific weight of evidence classifications for the nine carcinogens of concern are provided in Section 2.3.2.[14]

2.3.2 Health Effects of Individual Compounds

A discussion of the adverse health effects and the weight evidence classification for the nine carcinogens is presented below.

2.3.2.1. Benzene. Benzene administered orally to rats has resulted in increased incidences of Zymbal gland carcinomas.[15] In mice, inhalation exposures have shown subsequent anemia and other disorders of the blood forming tissues.[16] Other studies with mammalian cells have shown cytogenic abnormalities following benzene exposure.[17]

In humans, chronic exposure to benzene has resulted in abnormalities of the blood such as anemia, leucopenia, thromobocytopenia (pancytopenia). Epidemiological studies have shown highly statistically significant causal associations between leukemia and occupational exposure to benzene and benzene-containing solvents.

Other studies of human populations exposed to benzene have shown significant increases in chromosomal aberrations. In some instances, the aberrations have persisted for years after the cessation of exposure.[18]

According to IARC, there is sufficient evidence that benzene is a human carcinogen and limited evidence that it is carcinogenic in experimental animals.[19] EPA classifies benzene as a Group A carcinogen, a human carcinogen. The Group A classification is used only when there is sufficient evidence from human studies to support a causal association between exposure to a given substance and induction of cancer.

TABLE 2-3. CLASSIFICATION OF EVIDENCE BASED ON ANIMAL AND HUMAN DATA[a,b]

	Animal Evidence				
Human evidence	Sufficient	Limited	Inadequate	No data	No evidence
Sufficient	A	A	A	A	A
Limited	B1	B1	B1	B1	B1
Inadequate	B2	C	D	D	D
No data	B2	C	D	D	E
No evidence	B2	C	D	D	E

[a]The above assignments are presented for illustrative purposes. There may be nuances in the classification of both animal and human data indicating that different classifications than those given in the table should be assigned. Supporting data (e.g., structure-activity relationships, short-term test findings, etc.) should also be considered in the weight-of-evidence classification.

[b] A = human carcinogen
B2 = probable human carcinogen
C = possible human carcinogen
D = not classifiable as to human carcinogenicity
E = evidence of noncarcinogenicity for humans

2.3.2.2 Carbon tetrachloride. Carbon tetrachloride causes damage to the liver, lungs, kidneys and central nervous system in humans.[20] The potential for embryotoxicity exists, especially for males. The carcinogenicity of carbon tetrachloride has been observed in tests with three animal species. Primarily, the tumors found in the animals were liver tumors.[21] The human data on carcinogenicity of carbon tetrachloride are centered on case reports and one epidemiological study. Using the EPA weight of evidence criteria for carcinogenicity, carbon tetrachloride is classified as a probable human carcinogen, Group B2.

2.3.2.3 Chloroform. Exposure to chloroform has been associated with adverse effects on the liver, kidneys, and central nervous system of humans.[22] Additional effects on the human cardiac system have also been reported, including cardiac arrhythmias and cardiac arrest.[23] There is also some evidence that chloroform has carcinogenic potential in several animal species, including mice (eight strains), rats (two strains) and one strain of dogs. In these studies, chloroform was administered orally. The evidence for carcinogenicity of chloroform in animals includes statistically significant increases in kidney tumors in rats and mice, and liver tumors in mice.[24]

No epidemiological studies have been found evaluating chloroform by itself. But several studies have indicated small, but statistically significant increases in rectal, bladder and colon cancer in humans consuming drinking water that contained chloroform as well as other trihalomethanes. Because chloroform was not thought to be the only possible carcinogen in the drinking water, the studies cannot be used to define chloroform's carcinogenic potential in humans. At this time, the epidemiologic evidence for the carcinogenicity of chloroform is inadequate.[25] The overall weight of evidence classification for chloroform is B2--probable human carcinogen,[26] based on existing sufficient animal evidence and inadequate epidemiological evidence.

2.3.2.4 Ethylene dichloride. The adverse effects of ethylene dichloride (EDC) that have been reported in the literature are largely associated with the gastrointestinal and nervous systems in humans. Subtle neurological effects (e.g., fatigue, irritability, sleeplessness) may be

more prevalent than overt symptoms of central nervous system toxicity at lower concentrations.

EDC was shown to be carcinogenic in a National Cancer Institute lifetime bioassay. Several types of tumors were observed in both rats and mice. In rats, carcinogens of the forestomach and circulatory system hemangiosarcomas were found. Hepatocellular carcinomas, alveolar/bronchiolar adenomas, and mammary carcinomas were seen in mice exposed to EDC.[27] The route of exposure for this bioassay was gavage (oral). No statistically significant increases in tumors occurred in rats or mice following lifetime inhalation exposure.[28] No case reports on studies in humans concerning carcinogenicity of EDC were found in the literature.[29]

The weight of evidence classification for EDC is B2, meaning it is a probable carcinogen in humans.[30] The classification is based on sufficient animal evidence from the lifetime oral exposure bioassay along with an absence of epidemiologic data.

2.3.2.5 Methylene chloride. Bioassays conducted by the National Toxicology Program (NTP) demonstrated that methylene chloride is oncogenic (tumor-causing) in both rats and mice when exposed via inhalation. In the mouse bioassay, statistically significant increases in liver and lung tumors were observed.[31] Statistically significant increases in benign mammary gland tumors were seen in the rat bioassay.[32]

Data on humans exposed to methylene chloride, primarily in the workplace, are judged to be inadequate for evaluating the carcinogenic potential of methylene chloride. Therefore, methylene chloride is classified as a Group B2 carcinogen--probable human carcinogen--because there is sufficient animal evidence and inadequate epidemiological evidence.

There has been some difference of opinion on the carcinogenic potential of methylene chloride as related to species differences in metabolic pathways. The EPA has evaluated the latest data related to the risk of cancer and exposure to methylene chloride. The EPA has concluded that the evidence of the carcinogenic mechanism of methylene chloride and species differences in use of the metabolic pathways are not sufficient to support an estimate of zero cancer risk to humans.[33]

2.3.2.6 Perchloroethylene

In humans, transient liver damage has been linked to short-term exposures to perchloroethylene at relatively high levels. Some slight effects on the central nervous system have been reported in humans exposed to relatively high perchloroethylene concentrations.[34] Excluding carcinogenicity, toxicity testing in experimental animals, along with limited human data, suggest that long-term exposure to low concentrations of perchloroethylene is not likely to present a health concern.[35]

However, inhalation bioassays conducted by the National Toxicology Program on rats and mice of both sexes showed evidence of carcinogenicity for perchloroethylene.[36] In the National Toxicology Program studies, increases in mononuclear cell leukemia, and rare kidney tumors were observed in rats. Liver tumors were observed in mice.[37] Using the EPA weight of evidence classifications, perchloroethylene is considered a probable human carcinogen, Group B2.

2.3.2.7 Trichloroethylene. The evidence for carcinogenicity of trichloroethylene is shown by tumor induction in male rats and both sexes of mice by oral and inhalation exposure.[38] Statistically significant increases in renal adenocarcinomas and adenomas were observed in bioassays of male rats by either inhalation or oral exposures. Either exposure route produced elevated incidences of leukemia in one strain of male rats.[39] Inhalation exposure produced hepatomas and hepatocellular carcinoma (liver tumors) in two mouse strains. Inhalation exposure also produced malignant lymphomas in one strain of female mice.[40] Leydig cell tumors have also been reported in two studies.[41]

Epidemiological evidence for carcinogenic potential of trichloroethylene is inadequate. EPA reviewed seven epidemiologic studies or surveys and concluded all were inadequate to allow characterization of carcinogenic potential.[42]

EPA has classified trichloroethylene as a Group B2--probable human carcinogen.[43] This classification is based on the existence of sufficient animal evidence and inadequate epidemiological evidence.

2.3.2.8 Vinylidene chloride. Metabolism of vinylidene chloride produces substances (metabolites) that exert adverse effects on the liver

and kidneys.[44] Eighteen animal studies were identified by EPA in the literature that provided information on the carcinogenic potential of vinylidene chloride. None of the studies that were conducted using the inhalation exposure pathway was conducted over the lifetime of the study animals.[45] In the single study that was judged to be adequate for assessing carcinogenic potential, statistically significant increases in kidney tumors were observed in one strain of male mice.[46]

There is no adequate epidemiologic evidence to assess the carcinogenicity of vinylidene chloride in humans.[47] Because there is limited animal evidence for carcinogenicity and inadequate evidence from epidemiological studies, EPA has classified vinylidene chloride as a Group C--possible human carcinogen.[48]

2.3.2.9 Vinyl chloride. In mice, exposure to vinyl chloride via inhalation has produced lung tumors, mammary carcinomas and angiosarcomas of the liver (malignant tumors). Cancer of the liver and other organs was also observed in rats exposed to vinyl chloride.[49]

In occupational exposures of humans, vinyl chloride disease is the name given to the total clinical syndrome associated with vinyl chloride exposure. The disease includes circulatory disturbances in the extremities (hands and feet), Raynaud syndrome, skin changes and changes in liver function.[50]

Other studies have shown chromosomal aberrations in the lymphocytes of humans occupationally exposed to vinyl chloride.[51] In addition, increased incidences of fetal loss have been associated with occupational exposure to vinyl chloride.[52]

Studies of humans exposed to vinyl chloride in the workplace have shown a causal relationship between the vinyl chloride exposure and development of cancer of the liver, brain and lung.[53] Angiosarcomas are rare tumors. Finding seven cases of these tumors in a single group of workers at one vinyl chloride plant is strong evidence of the carcinogenicity of vinyl chloride. Vinyl chloride is classified as a Group A carcinogen; a human carcinogen.

2.3.3 Cancer Risk Assessment

Based on the available information, EPA attempted to quantify the potential carcinogenic risks to the people exposed to MSW landfill emissions. However, unlike other source categories that have been regulated previously, the EPA was not able to quantify for this source category as one of the critical risk assessment parameters--toxics emissions rates. This was because the MSW landfill data base was generated by collecting available information from numerous sources which were not specifically designed to quantify toxics emissions rates. Although the data base of 931 facilities contained some data for the toxic constituent concentrations and the landfill gas emissions rates, there was no facility for which both values were known. Both the toxic constituent concentration and the landfill gas emission rate are required to compute the toxic emission rate. In addition, the EPA had no reliable technique to replace a missing value for either toxics emissions parameter from the other reported parameters in the database. Thus, the EPA could not reliably calculate a toxics mass emission rate and, in turn, could not reliably calculate a risk estimate for even one facility.

Other attempts, such as random assignment of known (measured) values to those facilities with missing values, were made to extrapolate nationwide risk estimates from the limited toxics data.[54] In doing so, this extrapolation (the estimation of the toxic landfill gas mass emission rates) was creating an additional level of uncertainty above and beyond a more typical risk assessment. After considering this additional uncertainty in conjunction with the other known uncertainties associated with risk assessment, the EPA concluded that MSW landfill risk estimates would not be credible. Furthermore, because these regulations are being proposed under Sections 111(b) and (d) of the Clean Air Act and are technology-based, risk estimates were not required in selecting among the regulatory options (see Chapter 5). However, even though the risk associated with exposure to landfill emissions could not be reliably quantified, the available information indicates that toxic emissions do emanate from MSW landfills and suggests a need to regulate this source category's emissions.

2.4 METHANE EMISSIONS CONTRIBUTING TO GLOBAL WARMING

Greenhouse gases serve to trap heat from the sun and maintain the earth's climate. Methane and other greenhouse gases such as carbon dioxide and nitrous oxide occur naturally in the atmosphere. They serve as a thermal blanket allowing solar radiation to pass through the atmosphere while absorbing some of the infrared radiation emitted back from the earth's surface. The absorption of radiation warms the atmosphere and provides the present climate. The earth would be approximately 30 degrees colder without the presence of greenhouse gases. The atmospheric temperature will increase if the concentrations of greenhouse gases are increased.[55-57]

Anaerobic decomposition of municipal solid waste in landfills results in the decomposition of municipal solid waste in landfills results in the generation of methane and carbon dioxide. An estimate of the amount of methane and carbon dioxide from MSW landfills is provided in Chapter 3. Methane is more potent than CO_2 due to its radiative characteristics and other effects methane has on atmosphere chemistry. Molecule-for-molecule methane traps 20-30 times more infrared energy in the atmosphere. Therefore even a small increase in the methane concentration in the atmosphere is a concern to scientists trying to predict the warming of the climate.[58-60]

There is considerable uncertainty with regard not only to the timing but also to the ultimate magnitude of any global warming. However, there is currently strong scientific agreement that the increasing emissions of greenhouse gases such as methane will lead to temperature increases. Within EPA and the international scientific community efforts are underway to reduce these uncertainties, estimate the cost of mitigation, and identify possible control options. Reduction of methane emissions from MSW landfills is one of many options available to reduce possible global warming.

2.5 EXPLOSION HAZARDS

2.5.1 Health Effects Associated with the Explosivity Of Municipal Solid Waste Landfill Air Emissions

Decomposition of the waste in MSW landfill air emissions produces the explosive methane gas. If the methane accumulates in structures on or off-site, explosions or fires can result. MSW landfill air emissions have

resulted in documented explosions and fires both within and beyond the landfill boundaries. Section 2.5.2 of this chapter describes the welfare effects such as the explosion hazards and associated property damage. This section briefly presents information on the health effects associated with the explosions resulting from MSW landfill gas emissions.

MSW landfill gas can migrate off-site and emissions can escape into confined spaces such as basements, crawl-spaces, utility closets and false ceilings. Explosions of the gas have caused severe personal injury and death. Table 2-4 lists documented cases of acute injury and death caused by explosions and fires related to MSW landfill gas emissions.

2.5.2 Explosivity of MSW Landfill Air Emissions

MSW landfill gas is composed largely of methane and carbon dioxide. Methane gas is odorless and is highly explosive when mixed with air at a volume between 5 and 15 percent (the lower and upper explosive limits of methane). Methane can migrate off-site from the landfill and possibly collect in basements or crawl spaces of nearby structures. For example, methane has migrated from the Port Washington landfill in New York into homes near the landfill. Within two years, four explosions occurred in homes very near the landfill. Subsequent testing by the Nassau County Fire Marshall discovered explosive levels of methane in or around twelve homes in the vicinity of the landfill.

Table 2-5 lists documented examples of explosions or fires associated with MSW landfill gas. These examples show clearly that structural damage and the loss of facility use are real possibilities related to these gas explosions. Instances of facility abandonment are also documented as shown in the table.

There is also documentation that the presence and migration of MSW landfill emissions adversely affects property value of surrounding land parcels. For example, at the Midway landfill in the Seattle, Washington, area, MSW landfill gas migrated under a major interstate highway and percolated up in residential areas. There was immediate concern in the neighborhood; 11 families were evacuated.[61] A program to subsidize the sale of houses in the area was started by the City. Information collected

TABLE 2-4. ACUTE INJURY AND DEATHS ASSOCIATED WITH MUNICIPAL LANDFILL AIR EMISSIONS EXPLOSIONS AND FIRES[a]

Location, date	Incident, injury[b]
Comack, NJ. 1984	Gas migrated to landfill weigh-station on-site. Explosion killed one, injured one.
Manchester, NJ. 1983	Spark from landfill pump probably ignited gas. One person burned.
Cleveland, OH. 1980	Explosion at foundry adjacent to landfill. One killed.
Commerce City, CO. 1977	Explosion in tunnel being built under a railroad right-of-way. Two workmen killed, four fireman injured.
Sheridan, CO. 1975	Gas migrated into drainage pipe under construction. Welding truck led to fire. Two injured.
Sheridan, CO. 1975	Gas accumulated in drain pipe running through landfill. Children playing with candle caused explosion. Several children injured.
Richmond, VA. 1975	Gas migrated from nearby landfill into apartment. Two injured.
Winston-Salem, NC. 1969	Gas migrated from adjacent landfill into basement of armory. Lighted cigarette led to explosion. Three killed, five seriously injured.
Atlanta, GA. 1967	Gas migrated from adjacent landfill into sealed basement of single story recreation center building. Lighted cigarette led to explosion. Two workmen killed, six injured.
Madison, WI (no date given)	Explosion destroyed sidewall of a townhouse. Two people seriously injured.

[a]Reference 62.

[b]These incidences highlight explosions and health effects. Other incidences of explosions related to methane migration from MSW landfills and property destruction are given in Table 2-6.

TABLE 2-5. DOCUMENTED CASES OF LANDFILL GAS MIGRATION AND ASSOCIATED FIRES AND EXPLOSIONS[a]

Landfill name/location/date	Damages and other comments
Pittsburgh, PA September 1987	Offsite gas migration is suspected to have caused house to explode. Incident is under investigation. Toxics are being monitored in homes near the landfill.
Bakersfield Landfill Fresno, California April 1984	Fresno police bomb squad used site for practice. A bomb was buried and was detonated causing LFG explosion. Explosive levels of methane were also migrating off-site.
BKK Landfill West Covina, California August-October 1984	Twenty residences temporarily evacuated due to explosive methane levels in adjoining soils.
Babylon Landfill Comack, New Jersey May 1984	Methane migrated to a house on-site and exploded.
Hardy Road Landfill Akron, Ohio 1984	One house destroyed. Ten houses evacuated temporarily.
I-95 Landfill Lorton, Virginia 1984	Explosion and fire occurred.
Landfill near Lake Township Canton, Ohio 1984	Two homes and a day care center temporarily evacuated.
PJP Landfill Jersey City, New Jersey 1984	Landfill fires causing air pollution have been a continual problem.
Smithtown Landfill Smithtown, New York 1984	Explosion damaged room in transfer station.

(continued)

TABLE 2-5. (Continued)

Landfill name/location/date	Damages and other comments
Wallingford Landfill Wallingford, Connecticut June 1984	Explosive levels of methane detected in dog pound. Dog pound temporarily closed, ventilation system to be installed.
Anderson Township Landfill Cincinnati, Ohio 1983	Explosion destroyed residence across the street from the landfill. Minor injuries reported.
Monument Street Landfill Baltimore, Maryland April 1983	Vent pipes were not maintained causing vents to become nonfunctional. Street light fire was believed related to methane migration. Ongoing lawsuit concerns presence of priority pollutants.
Ocean County Landfill Manchester, New Jersey December 1983	Spark from landfill pump probably ignited methane gas, causing explosion and fire. Office building destroyed.
Operating Industries Landfill Monterey Park, California August 1983	Vinyl chloride detection caused SCAQMD to order 30-day shutdown of landfill. It reopened, subject to closure in 6 months.
Shawnee County Landfill Topeka, Kansas August 1983	Home abandoned due to high methane levels.
Fells Street Landfill Richmond, Virginia 1975	In 1975, explosion occurred in nearby apartment building. The city decided to buy and demolish it. Two schools sited on the landfill were closed until a control system was installed.
Tyler, Texas May 1982	TDPS office building sited on closed landfill. Methane has caused problems since early 1970's. Failure of ventilation exhaust fan resulted in "significantly high" levels of methane in the building.

(continued)

TABLE 2-5. (Continued)

Landfill name/location/date	Damages and other comments
Mission Avenue Oceanside, California 1981	Schools surrounding the landfill were evacuated and classes were suspended for 4-5 months.
Port Washington Landfill North Hempstead, New York 1981	Explosions in furnace rooms of several homes. Minor damage occurred. Furnaces were replaced.
Beantown Dump Rockville, Maryland 1980	Small explosion occurred in enclosed back room of auto body shop. Shop closed for 1 month until control system was installed.
Warner Hill Landfill Cleveland, Ohio 1980	Explosion killed foundry worker on site adjacent to landfill.
Reilly Construction Company Springfield, Illinois 1979	Methane migrated into construction company offices adjacent to the landfill. Limited fires occurred. No explosion. Building evacuated and use restricted for 4 weeks.
Allegheny County Landfill Frostburg, Maryland 1978	Limited fire in off-site equipment maintenance building. No explosion. Building use restricted for 2 months. Building was highly ventilated until gas control system installation.
Campground Landfill Louisville, Kentucky 1978	No physical damages occurred. Buildings evacuated for short period of time.
Lees Lane Landfill Louisville, Kentucky 1978	Small fires and explosions. Several houses evacuated and condemned. Benzene (29.5 ppm) and vinyl chloride (17.9-122.6 ppm) detected off-site.
Unnamed Landfill Adams County, Colorado	Explosion at a construction project adjacent to the landfill.

(continued)

TABLE 2-5. (Continued)

Landfill name/location/date	Damages and other comments
Fells Street Landfill Richmond, Virginia 1982	The 1982 incident occurred when children trespassed onto the landfill site, entered a control system manhole, and lit a match, resulting in an explosion.
Winston-Salem, North Carolina 1969	Methane migrated into National Guard Armory.
Greentree Hills Landfill Madison, Wisconsin	Explosion blew out one sidewall of a townhouse. Three adjacent apartment buildings and several homes evacuated for 20-30 days. Claims filed against the city total $5.2 million dollars.

[a]References 63,64.

on property values recorded during the operation of the program indicated a decrease of 5 to 10 percent in residential property value.

2.6 ODOR NUISANCE

Odors are frequently associated with air emissions from MSW landfills. Odors escape along with MSW landfill gas from surface cracks in the landfill. As waste is added to the landfill, disturbances of soil layers can also provide a means of escape for odors. Individuals vary in their ability to detect odors and in the degree of pleasantness or unpleasantness they experience with various odors.[65] However, the types of odors generally associated with the decomposition of organic material that occurs at landfills are most likely to be unpleasant or objectionable. This section describes the occurrence of odors at MSW landfills, and lists examples of the types of odorous compounds likely to emanate from landfills. The section also describes how odors affect human welfare by the unpleasantness of the odors themselves, by possibly lowering the property value of real estate near a MSW landfill, and by the potential for odors to cause properties to be abandoned and therefore leading to loss of facility and property use.

2.6.1 Odor Generation

Municipal landfill gas is generated largely by bacterial decomposition of organic materials in the municipal solid waste. As the decomposition proceeds, odiferous compounds can escape from the landfill through cracks in the landfill surface cover.

Other possible sources of odors associated with air emissions from MSW landfills are the actual wastes themselves. Household wastes that are often disposed in MSW landfills include chemicals in cleaners, paints, pesticides and adhesives. These consumer products often contain solvents or other compounds with distinctive odors. As these household products are added to a landfill, the odors associated with some of these chemicals may be noticeable to nearby residents or passersby. These odors may also emanate on a continuing basis from cracks in the landfill surface cover.

2.6.2 Adverse Effects of Odors on Human Welfare

The influence of odors on the comfort and welfare of individuals is difficult to prove. Odors can result in social and behavioral changes in an

exposed population. However, odor perception and impact is subjective. Different individuals may react differently to the same type and intensity of odor. Therefore, it is difficult to quantify a degree of unpleasantness associated with different odors. The descriptions in this section on the adverse effects of odors on human welfare are necessarily qualitative.

A few studies in the United States and the Federal Republic of Germany have investigated the social and behavioral effects of odors on the population. These studies have indicated that annoyance is a common reaction of residents in communities where unpleasant odors are encountered. Examples of responses from a survey of 704 residents of Dusseldorf are shown in Table 2-6. In the U.S., studies have indicated that odors have interfered with daily activities. U.S. studies are generally older and not quite as specific as other studies in the literature.[66]

It seems likely that the presence of odors would also exert the same type of detrimental effect on property value. At this time, the effect cannot be quantified. As was discussed earlier in relation to explosivity, property values around the Midway landfill in the Seattle area decreased from 5 to 10 percent with increased awareness of the presence of MSW landfill gas. The decreases could not be directly correlated to odors associated with the landfill.

Odors can also cause temporary or perhaps permanent loss of facility use. Although specific studies were not found that documented any loss of property use because of the odors from MSW landfills, it is possible that such adverse effects would occur. The responses shown in Table 2-7 indicate that odors can interfere in outdoor activities and interfere with the comfort of living. If a population perceives an odor as offensive and has questions about other possible effects beyond an annoying odor, the use of recreational or social facilities near the odor source may be greatly reduced or eliminated.

2.7 ADVERSE EFFECTS ON SOILS AND VEGETATION FROM MSW LANDFILL AIR EMISSIONS

The inability to grow vegetation or trees at MSW landfills is believed to be caused by one or more of the following factors: (1) lack of oxygen in the root zone; (2) toxicity of carbon dioxide to the roots; or (3) anaerobic

TABLE 2-6. RESPONSE COMPONENTS OF ANNOYANCE FACTORS DERIVED FROM A SURVEY OF 704 RESIDENTS OF DUSSELDORF[a]

Survey responses
Reduced social contacts
No pleasure in coming home
Odor leads to tensions within the family
Odor interferes with or disturbs communication
Odor spoils appetite
Odor interferes with comfort of living
Odor interferes with outdoor activities
Odor induces anger

[a]Reference 67.

conditions of the soil permitting the accumulation of reduced metals, such as iron (Fe), manganese (Mn) and zinc (Zn), in concentrations toxic to the vegetation.[68] Generally when landfill gases are present in the surface soil, the concentration increases at deeper soil layers. Thus, although the deeper rooted trees die, the shallow rooted ground vegetation continues to live. Diffusion of ambient air into the soil and diffusion of landfill gases out of the soil frequently result in the soils nearest the surface (top several inches) remaining in an aerobic condition, whereas the levels where the deepest roots are present can be anaerobic.[69]

According to the literature, there is a good deal of variability in tolerance to low oxygen in the root zone. The growth of red and black raspberries was inhibited by exposure to 10 percent oxygen, whereas apple trees required 10 percent oxygen in the soil in order to sustain growth.[70] Tomato plants grown in solution culture exhibited marked reduction in growth and ability to take up potassium (K) when exposed to three percent oxygen in the root zone.[71] Leone et al. reported that red maple, which is flood-tolerant, was also more tolerant of soil contaminated by simulated landfill gas than sugar maple, which is not tolerant of flooding.[72]

Greenhouse and field studies, and other research reported in the literature all confirm that the presence of landfill gases in the root zones of vegetation can be injurious to the extent of causing the death of vegetation. The major characteristics of landfill gas deleterious to plants when found in the root zone were the high carbon dioxide and methane and low oxygen concentrations resulting from anaerobic refuse decomposition.[73] Further studies indicate the extent of effects of landfill air emissions on vegetation. Various investigators have experienced difficulties in growing vegetation at completed or closed landfill sites. Stunting of corn and sweet potatoes became evident in areas adjacent to a New Jersey site where gases had migrated away from the landfill into the root zone of corn and sweet potato plants.[74,75] Death and poor growth of loblolly and other pines planted on such sites in southern Alabama have also been attributed to the presence of fermentation gases in the soil environment.[76] Poor tree growth in these areas has also been associated with lack of soil moisture and

increasing amounts of ammonia, nitrogen, iron, manganese, zinc, and copper.[77]

Closed refuse and fill sites at 15 locations in New Jersey, New York State, metropolitan New York City, New England, Washington, Oregon, and Alabama were sampled both where vegetation was dead or dying and where the same species were growing normally. Inspection of the contents of a soil sampling tube inserted to a depth of 20 cm in soil at sites where vegetation was dead or dying commonly revealed an anaerobic situation (dark, foul smelling soil). Soil at sites where plant species were growing were commonly found to be in aerobic conditions. Instrument readings of methane (CH_4) and carbon dioxide (CO_2) were as high as 50 percent and 43 percent respectively, at the anaerobic sites.[78]

Soil tests at a closed landfill in central New Jersey showed that, 4 years after closure, the deepest 15 cm of a 60 cm soil cover was still distinctly anaerobic. The upper 45 cm of soil showed evidence of aerobic conditions; however attempts to establish herbaceous vegetation at the site demonstrated that only a few grass species ["reliant" hard fescue (Festuca longifolia Thuil.), redtop (Agrostis alba L.) and sheep's fescue (F. ovina L.)] could survive under the undesirable soil conditions created by the landfill gases. Attempts to establish woody species also failed, even where grasses had been established.

Experimental work and site investigations have demonstrated an inability of the landfill cover to support and maintain vegetation, which also leads to increased erosion potential. If the cover is eroded, there is a chance that refuse will be exposed. Opening the landfill cover could lead to contaminated runoff from the site, increased odor nuisance, and increases in rodent or vermin populations. According to CFR Part 60, this may be defined as an effect on public welfare.

2.8 REFERENCES

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3. MUNICIPAL LANDFILL AIR EMISSIONS

This chapter presents a description of municipal solid waste landfills and a characterization of landfill air emissions. Section 3.1 provides an overview of municipal landfills. Section 3.2 describes the sources of emissions from municipal landfills, while Section 3.3 presents estimates of municipal landfill air emissions in 1992 (expected year of regulation) and projected emissions from new municipal landfills established between 1992 and 1997. Finally, Section 3.4 describes the explosion hazards and odor nuisance associated with municipal solid waste landfill air emissions.

3.1 GENERAL LANDFILL INFORMATION

The term "municipal solid waste landfill" in this document refers to landfills regulated under a subsection of Subtitle D of the Resource Conservation and Recovery Act (RCRA) that receive primarily household and/or commercial waste. RCRA Subtitle D landfills receive only nonhazardous waste (with the exception of small quantity generator and household hazardous waste) and are categorized according to the primary type of waste received. Municipal landfills may receive small quantities of waste types other than household and commercial wastes (as discussed in Section 3.1.1).

Based on the 1986 EPA survey of municipal solid waste landfills, there are presently an estimated 6,034 active municipal landfills in the United States receiving about 209 million megagrams (Mg) of waste annually. Of the 209 million Mg of waste received, approximately 150 million Mg (72 percent) is household waste and 58 million Mg (28 percent) is commercial waste. The total estimated design capacity of these active municipal landfills is 11,100 million Mg and the total estimated quantity of refuse in place is 4,330 million Mg. Thus, the overall proportion of total design capacity currently filled is about 39 percent.[1,2]

The distribution of landfill sizes based on design capacity and corresponding average refuse acceptance rates is shown in Table 3-1. Most of the active municipal landfills (about 93 percent) have a design capacity of 5 million Mg or less. The overall proportion of design capacity currently filled ranges from 29 percent for landfills having a design

TABLE 3-1. ACTIVE MUNICIPAL LANDFILL SIZE DISTRIBUTION[a]

Design capacity (million Mg)	Number of landfills	Percent of total landfill population	Percent filled[b]	Average acceptance rate (Mg/day)
<1	4,284	71	45	50
1-5	1,327	22	44	470
5-10	241	4	40	1,370
10-20	91	1.5	37	2,000
>20	91	1.5	29	3,910
TOTAL	6,034	100	-	-
Median			-	11.5
Average			39	282

[a]Reference 2

[b]Amount of refuse in place relative to the total design capacity of the landfill.

capacity greater than 20 million Mg to 45 percent for landfills having a design capacity less than 1 million Mg. The average acceptance rates range from 50 Mg/day for the smaller landfills (<1 million Mg design capacity) to about 4,000 Mg/day for the larger landfills (>20 million Mg design capacity).

There is a large difference between the average and median refuse acceptance rates for the total landfill population. This is due to the relatively small number of large municipal landfills which account for a disproportionately large share of the total waste received. The median value of annual refuse acceptance rate for the total landfill population is 3,000 Mg/yr (11.5 Mg/day), whereas the average value is 73,000 Mg/yr (282 Mg/day).[3]

3.1.1 Municipal Waste Composition

The types of waste potentially accepted by municipal landfills can be categorized into 12 waste types: (1) municipal solid waste, (2) household hazardous waste, (3) municipal sludge, (4) municipal waste combustion ash, (5) infectious waste, 6) waste tires, (7) industrial nonhazardous waste, (8) small quantity generator hazardous waste, (9) construction and demolition waste, (10) agricultural waste, (11) oil and gas waste; and (12) mining waste. The average composition of these waste found in municipal landfills is presented in Table 3-2. Below is a brief description of each waste type.

3.1.1.1 Municipal Solid Waste. Most of municipal solid waste (MSW) is comprised of paper and yardwastes. It is also comprised to a lesser extent of glass, metals, plastics, food wastes, rubber, textiles, and wood.[4]

3.1.1.2 Household Hazardous Waste. Household hazardous waste consists mostly of household cleaners, automotive products, home maintenance products, and lawn and garden products.[5]

3.1.1.3 Municipal Sludge. Municipal sludge is generated from drinking water and waste water treatment plants. Sewage sludge is predominantly organic matter, while drinking water sludge is a mixture of organic and inorganic components.

3.1.1.4 Municipal Waste Combustion Ash. This waste is derived from the incineration of municipal solid waste. About 90 percent of municipal

TABLE 3-2. AVERAGE COMPOSITION OF WASTE IN ACTIVE MUNICIPAL WASTE LANDFILLS

Waste type	Mean waste composition (wt %)
Household wastes	71.97
Commercial nonhazardous wastes	17.19
SQG Hazardous wastes	0.08
Asbestos-containing waste materials	0.16
Construction/Demolition wastes	5.83
Industrial process wastes	2.73
Infectious wastes	0.05
Municipal incinerator ash	0.08
Other incinerator ash	0.22
Sewage sludges	0.51
Other commercial wastes	1.19

waste combustion ash is currently disposed of in landfills. However, this practice may be prohibited in the future by EPA because of the concern that heavy metals present in combustion ash can be readily mobilized and transported in municipal landfill leachate. The EPA is currently conducting a study to determine the appropriate controls necessary for the management of municipal waste combustion ash.[6]

3.1.1.5 Infectious Waste. Infectious waste is by and large originated at hospitals and research testing labs. The types of infectious wastes include isolation wastes; cultures of infectious agents; human blood products; pathological wastes; contaminated injection needles; contaminated animal carcasses; and body parts and bedding.

3.1.1.6 Waste Tires. This waste includes discarded vehicle tires which eventually are deposited in a municipal landfill. It has been estimated that about 70 percent of discarded tires are disposed of in landfills.[7]

3.1.1.7 Industrial Non-Hazardous Waste. This category includes any refuse from industrial facilities that are not defined as hazardous waste under RCRA. Approximately 80 percent of this waste is generated by the following industries[8]: Industrial Organic Chemicals; Iron and Steel Manufacturing; Fertilizer and Agricultural Chemicals; Electric Power Generation; and Plastics and Resins Manufacturing.

3.1.1.8 Small Quantity Generator Hazardous Waste. Small quantity generators are defined in RCRA as those producing less than 100 kg per month of hazardous wastes. The dominant SQG waste type is used lead-acid batteries, comprising about 60 percent of SQG waste. The next most abundant SQG waste is spent solvents, comprising about 18 percent of SQG waste.[9]

3.1.1.9 Construction and Demolition Waste. Construction and demolition wastes consist mostly of concrete, asphalt, brick, stone, plaster, wallboard, glass, and piping. Paint and solvent waste associated with construction is considered a SQG waste.

3.1.1.10 Agricultural Wastes. Agricultural waste consist primarily of animal, crop, and irrigation wastes.

3.1.1.11 Oil and Gas Wastes. Oil and gas wastes are chiefly liquid brines and drilling muds.

3.1.1.12 Mining Wastes. Mining wastes are mostly debris from crushing, cleaning, and floatation processes used in the mineral extraction industry.

3.1.2 Landfill Design and Operation

The two major approaches to the design and operation of a municipal landfill are: 1) the trench method; and 2) the area method. The trench method involves excavating daily trenches designed to receive one day's waste. Daily trenches are typically 100 to 400 ft long, 3 to 6 ft deep, and 15 to 25 ft wide. The waste is spread in layers, 1.5 to 2 ft thick, and then compacted before the next layer is applied. The trench method is most suitable on flat or gently rolling land with a low ground water table.

The area method involves the application of waste over the natural ground surface. Waste is generally applied in layers of less than 2 ft and is then compacted before successive layers are applied. The area method is often used in areas such as California, where natural depressions (e.g. canyons) are abundant. If the landfill site has a high water table, the excavation method may not be feasible and the area method must be used.

Common to both landfilling methods is the basic landfill cell. A schematic of the cell design is provided in Figure 3-1.[10] A cell is usually designed to receive one day's waste and is closed at the end of the day. The height of the cell is usually less than 8 ft. The working face of a cell can extend to the facility boundaries. The waste is compacted into the cell at compaction densities range from 500 to 1500 lb/per cubic yard. After compaction, daily cover material is applied. Most states require that at least a 6-inch cover be applied at the end of the day. A 2-ft final cover of material capable of supporting vegetation is required for a completed landfill. Interim cover requirements (for areas unattended for a period of time) vary from State to State but are usually about 1 ft. After compaction and cell closure, settlement occurs. Ninety percent of the settlement occurs within the first five years.[11]

Liners can be used to prevent water entry to control leachate production. They are also used to control landfill gas migration. There are two basic types of landfill liners: soil and synthetic. Soil liners consist of compacted clay. These liners can achieve reduced permeabilities

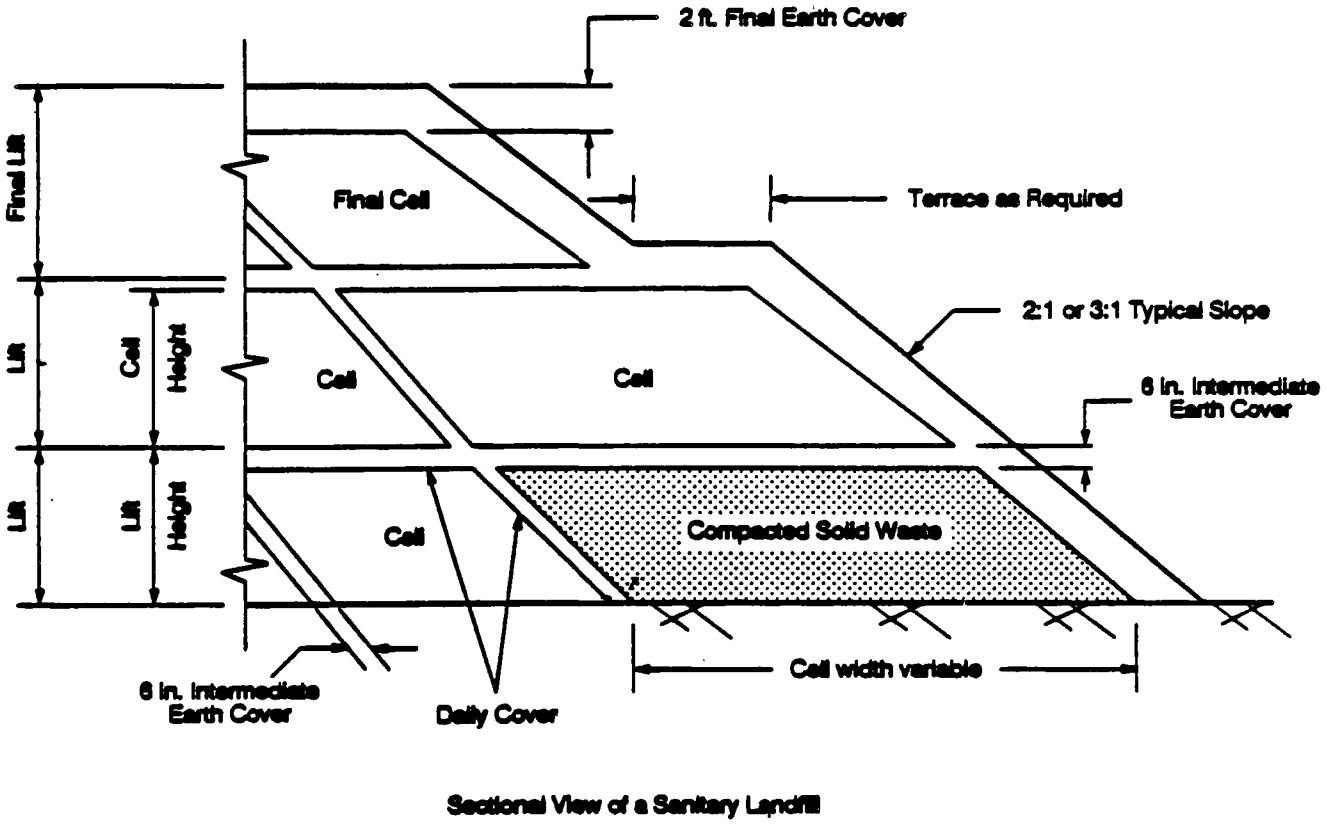


Figure 3-1. Landfill cell design.

of 10^{-7} cm per second. Types of synthetic liners include asphalt, cement, soil sealants, sprayed liquid rubbers, and synthetic polymeric membranes. Reduced permeabilities of 10^{-10} cm per second can be achieved with these liners.[12] Less than 16 percent of municipal landfills use liners. Clay liners are used more often than synthetic liners in Subtitle D landfills.[13]

Due to the Federal and State restrictions on acceptance of liquids in landfills, solidification or fixation of liquid waste is often required before it can be disposed. Liquid received in drums is decanted into pits or directly into the landfill. The liquid may be mixed with other waste, absorbents, or cement. Sometimes "trenching" or "lagooning" methods are used. These involve pouring liquids into excavated areas within a waste layer.[14] The liquid is then allowed to infiltrate downward and laterally to be absorbed by the waste. As reported in the summary of the 1986 EPA survey of MSW landfills, only 1.2 percent accept free liquid solvents, 5.4 percent accept bulk liquids, and 3 percent accept drummed liquids.[15] These percentage may vary in some parts of the U.S. However, prior to the Hazardous and Solid Waste Amendments of 1984, hazardous wastes (solids and liquids) were codisposed with municipal wastes at some landfills.[16]

3.2 EMISSIONS FROM MUNICIPAL LANDFILLS

This section is divided into five subsections. The source of municipal landfill air emissions is identified in Section 3.2.1. The mechanisms responsible for these emissions are discussed in Section 3.2.2. The factors impacting the emissions mechanisms, and thus emission rate, are discussed in Section 3.2.3. Reported emission rates and a technique for estimating emissions from municipal landfills are presented in Section 3.2.4. Finally, Section 3.2.5 discusses landfill gas composition.

3.2.1 Landfill Cells

Landfill cells represent the major source of volatile constituents from municipal landfills. As the waste is received, it is initially placed in an open cell. At this time, the waste is in direct contact with the ambient air and some loss of volatile constituents to the atmosphere is likely. This newly received waste is likely to remain in contact with the ambient air for a period of several hours as the waste is covered and possibly

compacted with other newly received wastes. In good practice, a 6-inch soil cover is placed over the newly received wastes at the end of the day. However, emissions of volatile constituents continue (through the soil cover) as the landfill cell is completed and after the landfill cell is closed. In addition to emissions of nonmethane organic compounds (NMOC) contained in the waste placed in the landfill cell, volatile organics may also be produced by biological processes or chemical reactions in the landfill as well.

Many municipal landfills are equipped with landfill gas collection systems. The purpose of these collection systems is to vent or collect landfill gas generated from the biological degradation of municipal type wastes. The two basic types of collection systems employed are: passive and active. Passive collection systems are generally installed to vent landfill gases to the atmosphere for the purpose of preventing lateral migration or to reduce the potential for explosion. Although the vented gas composition is primarily methane and CO_2, of NMOC are also present in the vented landfill gas.[17]

Active collection systems include blowers or compressors and are generally vented to a flare or energy recovery equipment (e.g., boiler, gas turbine, internal combustions engines). However, active collection systems may also be vented directly to the atmosphere. Even at landfills with flares or energy recovery equipment, these systems may be a significant emission source. During periods of equipment malfunction, the collected landfill gas is often discharged directly to the atmosphere. In addition, the objective of energy recovery systems is to recover methane from the landfill gas stream. As part of the recovery scheme, nonmethane constituents may be removed and discharged to the atmosphere.

3.2.2 Landfill Emission Mechanisms

Mechanisms governing the rate of organic emissions from landfill cells can be separated into two types: production and transport. For emissions to occur, the volatile organic must first be present in gaseous form. The gaseous organic compound must then be transported to the atmosphere above the landfill. Either mechanism can limit the emission rate. However, transport appears to be the limiting emission mechanism.

3.2.2.1 Production Mechanisms. The first step governing municipal landfill air emissions is the production of the pollutant in its vapor phase. This may be accomplished through one of three production mechanisms: vaporization, biological decomposition, or chemical reaction.

3.2.2.2 Vaporization. Vaporization is the change of state from liquid or solid to vapor. The change of state occurs due to the chemical phase equilibrium that exists within the landfill. Organic compounds in the landfill cell will vaporize until the equilibrium vapor concentration is reached.

3.2.2.3 Biological Decomposition. A second mechanism by which a volatile constituent may be produced in its vapor phase is biological decomposition. Higher molecular weight organic constituents in the landfill wastes may be decomposed by naturally occurring bacteria. The product of this decomposition can be a lower molecular weight constituent with a higher vapor pressure or volatility. For example, vinyl chloride is formed as a result of degradation of trichloroethene and dichloroethene in the refuse.[18] It has also been suggested that lignin in municipal waste forms substituted aromatics and eventually forms benzene, toluene, phenols, alcohols, ketones, and esters.[19]

The production of volatile organics (other than methane) is dependent on the availability of nutrients for bacteria, refuse composition, moisture content of the waste, oxygen availability, age of landfill, the presence of biological inhibiters, temperature, and pH.

3.2.2.4 Chemical Reaction. The chemical reaction of materials present in landfills is another possible mechanism for the production of volatile constituents. These reactions may occur as the result of contact between reactive wastes placed in the landfill or reactive gases generated in the landfill.

3.2.2.5 Transport Mechanisms. When a volatile constituent is present in its vapor phase, it can be transported to the surface of the landfill, through the air boundary layer above the landfill, and into the atmosphere. This transport may occur all or in part by one of three major transport mechanisms: diffusion, convection, and displacement. Diffusion can be further broken down into molecular diffusion through pores in the

landfill and diffusion through the air boundary layer above the landfill. Displacement can be further broken down into displacement due to compaction and settlement of the waste, displacement due to barometric pressure changes, and displacement due to ground water table fluctuations.

For municipal landfills, landfill gas convection is by far the predominant transport mechanism. Landfill gas, mainly consisting of methane and carbon dioxide produced by the biodegradation of refuse, sweeps vapors present in the landfill to the landfill surface as it flows through the refuse. The generation of landfill gas is discussed in detail in Section 3.2.3.

3.2.3 Factors Affecting Municipal Landfill Air Emissions

As discussed in the previous section, municipal landfill emission rates are a function of production and transport mechanisms. Either mechanism can be the rate determining mechanism. However, transport appears to be the limiting one.

3.2.3.1 Factors Affecting Production Mechanism. As discussed in the previous section, there are three types of production mechanisms active in landfills: vaporization, chemical reaction, and biological decomposition. The factors affecting each of these production mechanisms are summarized in Table 3-3.

As shown in Table 3-3, the major factors affecting vaporization are the concentration of individual compounds in the landfill, physical properties of the individual organic constituent, and the landfill conditions. The emission rate of a specific organic constituent is expected to be a direct function of its concentration in the landfill. Assuming that vaporization is controlled by equilibrium rather than kinetics, the physical properties important to the rate of vaporization are the pollutant vapor pressure, solubility in water, and partition coefficient between the adsorbed and free phases. Compounds with higher vapor pressure to solubility ratios (pseudo Henry's Law Constant) vaporize faster. Also, adsorption of the organic constituent onto solids present in the landfill can play a key role in determining the equilibrium concentration of the organic constituent. The octanol-water coefficient of the organic compound is an indicator of the partitioning between the adsorbed and free phases. Compounds with lower

TABLE 3-3. FACTORS AFFECTING PRODUCTION MECHANISMS

Mechanism	Factors affecting mechanism
Vaporization	- Partial pressure of the constituent
	- Constituent concentration at the liquid-air interface
	- Temperature
	- Confining pressure
Chemical reaction	- Composition of waste
	- Temperature
	- Moisture content
	- Practice of separate disposal areas for different waste types
Biological decomposition of liquid and solid compounds into other chemical species	- Nutrient availability for bacteria
	- Refuse composition
	- Age of landfill
	- Moisture content
	- Oxygen availability
	- Industrial waste acting as biological inhibitors (toxic to bacteria)
	- Temperature
	- pH

octanol-water coefficient are adsorbed onto organic solids less readily and tend to vaporize more quickly. Other factors that can affect the rate of vaporization are the landfill temperature and pressure. Higher temperatures and lower pressure yield higher vaporization rates.

The extent to which chemical reactions lead to municipal landfill air emissions is not well understood. Obviously two incompatible (reactive) compounds must be present in the same location of the landfill in order for a chemical reaction to take place. The primary factor affecting the rate of production due to chemical reaction is the composition of the refuse placed in landfill cells. Possible chemical reactions are also affected by the landfill temperature, but only if the reactive compounds are present. Higher temperature can result in either increased or decreased reaction rates.

Biological decomposition of one organic compound into another is affected by the composition of the landfill refuse and the landfill conditions supporting biological activity. In order for a compound to be produced, a predecessor compound must first be present in the landfill. In addition, conditions in the landfill must be supportive of the particular bacteria responsible for the decomposition. Bacteria present in landfills are in general sensitive to nutrient availability, age of the refuse, moisture content, temperature, oxygen availability, biological inhibitors, and pH. The best overall indicator of biological activity is the rate of landfill gas generation, since landfill gas is the product of refuse decomposition.

3.2.3.2 Factors Affecting Transport Mechanism. As discussed previously, there are a number of transport mechanisms active in landfills. These include molecular diffusion, landfill gas convection, displacement due to compaction and settling, displacement due to barometric pressure changes, and displacement due to water table fluctuations. The factors affecting each of these identified transport mechanisms are summarized in Table 3-4. Although landfill gas convection is by far the major factor affecting the emission rate from landfills, factors affecting the other identified transport mechanisms are also discussed below.

TABLE 3-4. FACTORS AFFECTING TRANSPORT MECHANISMS

Mechanism	Factors affecting mechanism
Molecular diffusion through soil cover	- Soil porosity - Concentration gradient - Diffusivity of constituent - Soil thickness
Molecular diffusion through boundary layer	- Wind speed - Concentration gradient - Diffusivity of constituent
Biogas convection	- Nutrient availability for bacteria - Refuse composition - Moisture content - Age of landfill - Oxygen availability - Industrial waste acting as biological inhibitors - Temperature - pH - Presence of gas collection system
Displacement due to compaction and settlement	- Amount of compaction practiced - Compatibility of waste - Overburden weight (settlement)
Displacement due to barometric pressure changes	- Changes in atmospheric pressure
Displacement due to water table fluctuations	- Rate of precipitation - Rate of evaporation - Horizontal versus vertical permeability - Presence of a liner

Molecular diffusion is the transport of a volatile organic due to the concentration gradient existing between a point in the landfill and the ambient air above. Factors affecting the rate of molecular diffusion include the concentration gradient, the diffusivity of the organic compound, the porosity of the soil cover, the cover thickness, and the wind speed above the landfill. The most important factor affecting the rate of diffusion is the concentration of organics in the landfill vapor, since the concentration of organics in the ambient air is relatively low (compared to landfill concentrations). In addition, the rate of diffusion is directly affected by the diffusivity of the organic compound, the soil cover thickness, and the soil cover porosity. The gas phase transport above the landfill is also affected by wind speed. Higher wind speeds reduce the width of the concentration gradient and thus increase the rate of diffusion.

The emission rate due to displacement mechanisms is directly affected by the volume of gas displaced. Higher compaction densities result in higher emission rates due to compaction. Highly variable barometric pressures result in higher emission rates due to barometric pumping, and highly variable water table levels result in higher emission rates due to water table fluctuations.

Among the different types of transport mechanisms, landfill gas convection is the predominant transport mechanism. In addition, landfill gas generation is also an indicator of biological activity in the landfill, and should indicate the production rate of organics due to biological decomposition.

3.2.4 Landfill Air Emissions Rate

Landfill gas, consisting primarily of methane and carbon dioxide, is produced by microorganisms in the landfill under anaerobic conditions. Anaerobic decomposition of complex organic material is normally a two-state process as shown in Figure 3-2.[20] In the first stage, there is no methane production. The complex organics are altered in form by a group of facilitative and anaerobic bacteria commonly called "acid formers". Complex materials such as cellulose, fats, proteins, and carbohydrates are hydrolyzed, fermented, and biologically converted to simple organic materials. Usually, the end products of the first stage are organic fatty

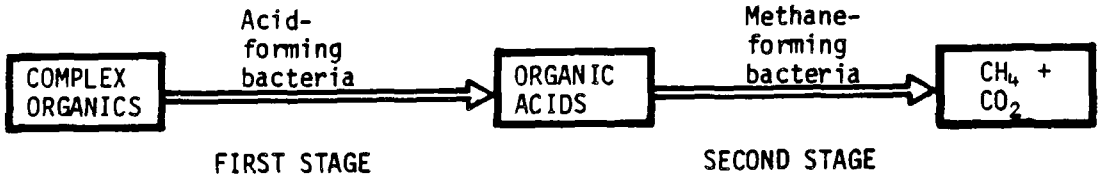


Figure 3-2. Two stages of anaerobic decomposition of complex organic wastes.20

acids. During the second stage of methane fermentation, the organic acids are consumed by methanogenic bacteria and converted into methane and carbon dioxide. The methanogenic bacteria are strictly anaerobic, and even small quantities of oxygen are toxic to them.[21]

3.2.4.1 Factors Affecting Landfill Gas Generation Rate

Landfill gas generation rate is a function of the following factors:

- Composition of refuse,
- moisture content of refuse,
- age of refuse,
- temperature of the landfill,
- pH and alkalinity of the landfill, and
- quantity and quality of nutrients.

3.2.4.2 Composition of Refuse. Refuse composition directly affects the rate of landfill gas generation. The higher the percentage of biodegradable materials (e.g., food and garden wastes, paper, textiles, and wood), the higher the landfill gas generation rate. Refuse composition can change with seasons and geographical locations. For example, there is higher percentage of garden wastes in tropical or fast-growing geographic areas. Certain compounds potentially present in the waste may be toxic to any bacteria active in the landfill and can upset the activity of methanogenic bacteria, resulting in a decreased gas generation rate. Examples of such substances are toxic organic solvents like carbon tetrachloride, chloroform and common salts of sodium, potassium, magnesium, calcium, ammonium, and sulfide at high concentrations.[22]

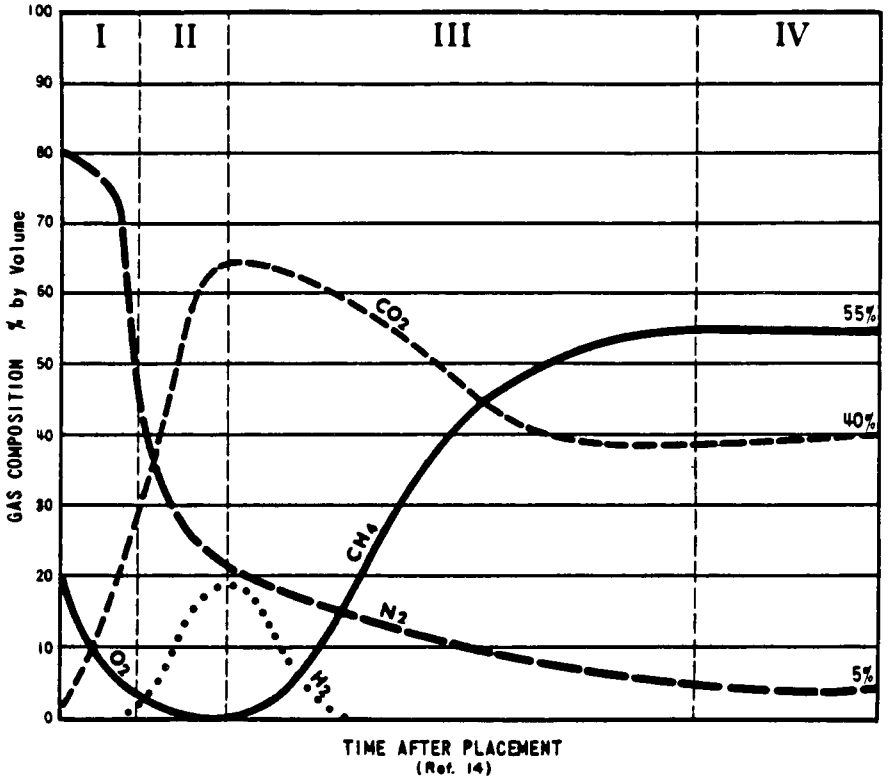
3.2.4.3 Moisture Content of Refuse. A high refuse moisture content (60 to 90 percent, wet weight basis) can increase the landfill gas generation rate. However, a typical refuse moisture content at the time of placement is about 25 percent. Since landfill design and operation usually focus on preventing water entry to control leachate production, landfill moisture content usually remains low.

3.2.4.4 Age of Refuse. Landfill gas generation rate and composition go through different phases throughout the lifetime of a landfill. The changes in gas composition can be characterized by four distinct phases (Figure 3-3).[23] In the first phase (several days to weeks), oxygen is present from the time of waste placement and carbon dioxide is the principal gas produced. In the second phase, an anaerobic condition exists once oxygen has been depleted. During this period, significant amounts of carbon dioxide and some hydrogen are produced. During the anaerobic third phase, methane production is initiated and the amount of carbon dioxide produced decreases. The fourth phase is also anaerobic in which gas production rate approaches pseudo-steady state. The duration of each phase is a function of the specific conditions within the landfill. Once methane production begins, it continues for a number of years (reportedly 17 to 57 years). The total time of gas generation depends on landfill conditions. For moderately decomposable wastes in a typical landfill, the gas generation rate peaks within six years after initial waste placement and declines steadily afterwards.[24]

3.2.4.5 Temperature of the Landfill. The methane production rate is sensitive to the landfill temperature. The optimum temperature for anaerobic digestion of refuse is 29°C to 38°C for mesophilic operation and 49°C to 57°C for thermophilic operation.[25] At temperatures below 10°C, there is a dramatic drop in generation rate.

3.2.4.6 pH of the Landfill. The optimal pH for methane fermentation is in neutral to slightly alkaline range (7.0 ~ 7.2). Initially, most landfills have an acidic environment for the first several years but the pH rises towards neutrality after those years.

3.2.4.7 Landfill Gas Generation Rate Model. Landfill gas generated by the methanogens acts as a stripping (or transport) gas for the nonmethane organic compounds (NMOCs) present in municipal landfills. Based on available data, the landfill gas production rate appears to range from 0.75 to 34 liters of landfill gas per kilogram of wet refuse per year.[26-28] As discussed in Section 3.2.4.1, there are several site-specific factors that affect the landfill gas generation rate. These factors cause the generation rate to be highly variable from landfill to landfill and



I. Aerobic

II. Anaerobic, Non-Methanogenic

III. Anaerobic, Methanogenic, Unsteady

IV. Anaerobic, Methanogenic, Steady

Figure 3-3. Evolution of typical landfill gas composition.[23]

difficult to predict. In an attempt to account for the site-specific conditions, a theoretical model and be used to predict the gas generation rate.

Several models are available for estimating the gas generation rate from a landfill using site-specific input parameters. Three relatively simplistic (first-order kinetic) models are the Palos Verdes, Sheldon-Arleta, and Scholl Canyon models. There are other models such as GTLEACH-I which treats the landfill as a fixed-film microbial treatment process operating in a batch-wise configuration with a continuous dilution and wash out. However, GTLEACH-I requires extensive input data which includes numerous initial concentrations, moisture content, and leachate flowrate.[29]

The basic approach in landfill gas generation modeling is to use the most simplified model available that is consistent with fundamental principals. The model is then empirically adjusted for the kinetic rate constant(s) to account for variations in refuse moisture content and other landfill conditions. The Scholl Canyon model which is a first order, single stage model was chosen to estimate the landfill gas generation rate for analyses presented in this document.[30] It is the most simplistic model with only two parameters and yields comparable results to other models, if comparable input values are used.

The Scholl Canyon model assumes that the gas production rate is at its peak upon initial waste placement, after a negligible lag time during which anaerobic conditions are established in the landfill. The gas production rate is then assumed to decrease exponentially (i.e., first order decay) as the organic fraction of the landfill refuse decreases. The Scholl Canyon model can be refined further by dividing the landfill into smaller submasses to account for different ages of the refuse accumulated over time. A convenient submass for computational purposes is the amount of refuse accumulated in one year. The total methane generation from the entire landfill (sum of each submass' contribution) is at its peak upon the landfill closure if a constant annual acceptance rate is assumed.

Assuming that the refuse has been accepted at the same annual rate over time (i.e. all submasses are of the same size), the model equation is as follow:

$$Q_{CH4} = L_0 R \{\exp(-kc) - \exp(-kt)\}$$

where,

Q_{CH4} = methane generation rate at time t, m^3/yr

L_0 = potential methane generation capacity of the refuse, m^3/Mg

R = average annual refuse acceptance rate during active life, Mg/yr

k = methane generation rate constant, 1/yr

c = time since landfill closure, year

t = time since the initial refuse placement, year

Lag time during which anaerobic conditions are established can be incorporated into the Scholl Canyon model by substituting c and t by (c + lag time) and (t + lag time), respectively. The typical lag time ranges from 200 days to several years depending on the landfill conditions.[31]

The theoretical value for potential methane generation capacity of refuse, L_0, depends on the type of refuse only. The higher the cellulose content of the refuse, the higher the value of the theoretical methane generation capacity. The theoretical methane generation capacity is determined by a stoichiometric method which is based on a gross empirical formula representing the chemical composition of composite refuse or individual refuse type. Some researchers have reported "obtainable L_0" which accounts for the nutrient availability, pH, and moisture content within the landfill. The researchers point out that "obtainable L_0" is less than the theoretical L_0. Even though refuse may have a high cellulose content, if the landfill conditions are not hospitable to the methanogens, the potential methane generation capacity of the refuse may never be reached. The "obtainable L_0" is approximated from overall biodegradability of "typical" composite refuse or individual waste components, assuming a conversion efficiency based on landfill conditions. The reported values of

theoretical and obtainable L_0 (along with the estimation method) range from 220 to 9540 ft^3 (6.2 to 270.1 m^3) CH_4 per Mg of refuse.[32-43]

The methane generation rate constant, k, determines how quickly the methane generation rate decreases, once it reaches the peak rate upon placement. The higher the value of k, the faster the methane generation rate from each submass decreases over time. The value of k is a function of the following major factors: (1) refuse moisture content, (2) availability of the nutrients for methanogens, (3) pH, and (4) temperature. In general, increasing moisture content increases the rate of methane generation rate up to a moisture level of 60 percent, above which the generation rate does not increase.[44] The pH of 6.6 to 7.4 is thought to be optimal for methanogens. Some studies suggest buffering to moderate the effects of volatile acids and other acid products which tend to depress the pH below the optimal pH.[45,46] Temperature affects microbial activity within the landfill, which in turn affects the temperature of the landfill. Warm landfill temperatures favor methane production and methane production may also reflect seasonal temperature fluctuation in cold climates where the landfill is shallow and sensitive to ambient temperatures. Values of k obtained from available literature, laboratory simulator results, industry experts, and South Coast Air Quality Management District (SCAQMD) test reports, and back-calculated from measured gas generations rates and Section 114 letter responses, and industry experts range from .003 to 0.21 1/yr.[47-51]

Other methods for estimating the nationwide NMOC emissions were evaluated. A comparison of these methods is provided in Appendix D. The alternative methods include the NMOC emission factor method, the South Coast Air Quality Management District method, and the municipal waste generation rate method.

3.2.5 Landfill Gas Composition

Landfill gas consists of approximately 50 percent by volume carbon dioxide, 50 percent methane, and trace amounts of nonmethane organic compounds (NMOC). The concentration of NMOC can range from 237 ppm to 14,294 ppm as shown in Table 3-5. The sources for the data provided in

TABLE 3-5. NMOC CONCENTRATIONS

Landfill ID	Co-disposal site	NMOC concentration (ppm)	Reference
A	No	237	52
B	No	244	53
C	No	364	54
D	No	487	55
E	No	514	56
F	No	528	57
G	Yes	595	58
H	No	639	59
I	No	704	60
J	Yes	710	61
K	No	947	62
L	No	1,060	63
M	No	1,066	64
N	No	1,135	65
O	No	1,356	66
P	No	1,372	67,68
Q	Yes	1,519	69
R	Yes	1,560	70
S	No	6,381	71
T	No	6,555	72
U	No	7,857	73
V	Yes	11,793	74
W	Yes	14,294	75

Table 3-5 are Waste Management of North America, South Coast Air Quality Management test reports, and responses to Section 114 questionnaires[52-75].

Concentrations of individual nonmethane organic compounds found in landfill gas are summarized in Table 3-6. After carbon dioxide and methane, ethane, toluene, and methylene chloride are the next major constituents in landfill gas with average concentrations of up to 143, 52, and 20 ppm, respectively. The most frequently detected compounds are trichloroethene, benzene, and vinyl chloride. These results are based on responses to Section 114 letters for 46 landfills.[76-81] Details for the 46 landfills are provided in Appendix C.

The organic air emissions from municipal landfills may include some toxic compounds and hazardous compounds with carcinogenic and other noncancer health effects. The carcinogenic and noncancer health effects resulting from exposure to these compounds are summarized in Chapter 2.

3.3 BASELINE EMISSION ESTIMATES

Baseline emission estimates are presented in this section for three categories of municipal solid waste landfills: existing active landfills, existing closed landfills, and new landfills. In this document, existing active landfills are defined as those landfills which receive municipal refuse prior to March 1, 1992 (the estimated promulgation date) and continue to receive municipal refuse. Landfill gas emissions are expected from the refuse already placed in these landfills as well as future refuse placements. The second category of landfills, existing closed landfills, are defined as those landfills which received municipal waste after November 7, 1987, but reached capacity and closed before March 1, 1992. Although no new refuse has been placed in these landfills since 1987, emissions will continue to evolve from these landfills until the refuse completely decays. The universe of closed landfills is much larger than defined here, but has been limited to this small subset due to the lack of information on the numerous landfills closed prior to 1987. The third category of municipal landfills, new landfills, is defined as those landfills which first receive municipal waste on or after March 1, 1992. The contribution of nationwide MSW landfill air emissions from these new

TABLE 3-6. SUMMARY OF NONMETHANE ORGANIC COMPOUNDS FOUND IN LANDFILL GAS[a]

CHEMICAL NAME	No. of Times Quantified	Average Conc. ppm	Average Conc. Detected ppm	Highest Conc. ppm	Lowest Conc. ppm
ETHANE	26	142.79	252.63	1780	0
TOLUENE	40	51.60	59.34	758	0.2
METHYLENE CHLORIDE	37	19.70	24.5	174	0
HYDROGEN SULFIDE	3	16.50	252.97	700	11
ETHYLBENZENE	31	14.64	21.73	428	0.15
XYLENE	2	14.52	333.85	664	3.7
1,2 - DIMETHYL BENZENE	1	12.78	588	588	588
LIMONENE	1	10.22	470	470	470
TOTAL XYLENE ISOMERS	27	10.04	17.11	70.9	0
α-PINENE	1	9.70	446	446	446
DICHLORODIFLUOROMETHANE	31	8.83	13.1	43.99	0
ETHYLESTER BUTANOIC ACID	1	8.65	398	398	398
PROPANE	26	7.68	13.59	86.5	0
TETRACHLOROETHENE	39	7.15	8.43	77	0
VINYL CHLORIDE	42	7.04	7.71	48.1	0
METHYLESTER BUTANOIC ACID	1	6.63	305	305	305
ETHYLESTER ACETIC ACID	1	6.13	282	282	282
PROPYLESTER BUTANOIC ACID	1	5.50	253	253	253
1,2 - DICHLOROETHENE	37	5.09	6.33	84.7	0
METHYL ETHYL KETONE	27	4.80	8.17	57.5	0
THIOBISMETHANE	1	4.57	210	210	210
METHLYCYCLOHEXANE	2	4.33	99.7	197	2.4
TRICHLOROETHENE	44	3.80	3.98	34	0.01
NONANE	1	3.63	167	167	167
BENZENE	45	3.52	3.6	52.2	0
ETHANOL	1	3.41	157	157	157
ACETONE	26	3.36	5.94	32	0
2 - BUTANOL	1	3.30	152	152	152
OCTANE	1	3.30	152	152	152

(continued)

TABLE 3-6. (Continued)

CHEMICAL NAME	No. of Times Quantified	Average Conc. ppm	Average Conc. Detected ppm	Highest Conc. ppm	Lowest Conc. ppm
PENTANE	26	3.19	5.64	46.53	0
HEXANE	26	3.01	5.33	25	0
METHYLESTER ACETIC ACID	1	2.96	136	136	136
1 - METHOXY - 2 - METHYL PROPANE	1	2.96	136	136	136
2 - BUTANONE	1	2.80	129	129	129
1,1 - DICHLOROETHANE	33	2.52	3.51	19.5	0
1 - BUTANOL	1	2.17	100	100	100
BUTANE	26	2.08	3.68	32	0
4 - METHYL - 2 - PENTANONE	1	1.93	89	89	89
2 - METHYL PROPANE	1	1.83	84	84	84
1 - METHYLETHYLESTER BUTANOIC ACID	1	1.50	69	69	69
2 - METHYL, METHYLESTER PROPANOIC ACID	1	1.50	69	69	69
CARBON TETRACHLORIDE	37	1.49	1.85	68.3	0
CHLOROETHANE	29	1.28	2.03	9.2	0
1,1,3 TRIMETHYL CYCLOHEXANE	1	1.24	57	57	57
2 - METHYL - 1 - PROPANOL	1	1.11	51	51	51
1,2 - DICHLOROETHANE	37	1.05	1.3	30.1	0
TRICHLOROFLUOROMETHANE	46	0.99	0.99	11.9	0
CHLOROMETHANE	30	0.90	1.38	10.22	0
2,5 DIMETHYL FURAN	1	0.89	41	41	41
2 - METHYL FURAN	1	0.87	40	40	40
CHLORODIFLUOROMETHANE	27	0.79	1.35	12.58	0
PROPENE	1	0.78	36	36	36
METHYL ISOBUTYL KETONE	26	0.78	1.38	11.5	0
ETHYL MERCAPTAN	3	0.78	11.93	23.8	1
DICHLOROFLUOROMETHANE	28	0.73	1.2	26.11	0
1,1,1 - TRICHLOROETHANE	38	0.69	0.84	9	0
TETRAHYDROFURAN	1	0.65	30	30	30
ETHYLESTER PROPANOIC ACID	1	0.57	26	26	26

(continued)

TABLE 3-6. (Continued)

CHEMICAL NAME	No. of Times Quantified	Average Conc. ppm	Average Conc. Detected ppm	Highest Conc. ppm	Lowest Conc. ppm
BROMODICHLOROMETHANE	29	0.45	0.71	7.85	0
ETHYL ACETATE	1	0.43	20	20	20
3 - METHYLHEXANE	1	0.43	20	20	20
C10H16 UNSATURATED HYDROCARBON	1	0.33	15	15	15
METHYLPROPANE	1	0.26	12	12	12
CHLOROBENZENE	29	0.24	0.38	10	0
ACRYLONITRILE	26	0.18	0.32	7.4	0
METHYLETHYLPROPANOATE	1	0.16	7.3	7.3	7.3
1,1 - DICHLOROETHENE	32	0.16	0.23	3.1	0
METHYL MERCAPTAN	3	0.12	1.87	3.3	1
1,2 - DICHLOROPROPANE	28	0.07	0.12	1.8	0
i - PROPYL MERCAPTAN	2	0.07	1.55	2.1	1
CHLOROFORM	36	0.06	0.08	1.56	0
1,1,2,2 - TETRACHLOROETHANE	28	0.06	0.1	2.35	0
1,1,2,2 - TETRACHLOROETHENE	2	0.06	1.33	2.6	0.05
2 - CHLOROETHYLVINYL ETHER	28	0.05	0.08	2.25	0
t - BUTYL MERCAPTAN	2	0.03	0.64	1	0.28
DIMETHYL SULFIDE	2	0.02	0.55	1	0.1
DICHLOROTETRAFLUOROETHANE	1	0.02	1.1	1.1	1.1
DIMETHYL DISULFIDE	2	0.02	0.55	1	0.1
CARBONYL SULFIDE	1	0.02	1	1	1
1,1,2-TRICHLORO 1,2,2-TRIFLUOROETHANE	1	0.01	0.5	0.5	0.5
METHYL ETHYL SULFIDE	1	0.01	0.32	0.32	0
1,1,2 - TRICHLOROETHANE	28	0.00	0	0.1	0
1,3 - BROMOCHLOROPROPANE	1	0.00	0.01	0.01	0.01
1,2 - DIBROMOETHANE	2	0.00	0	0	0
C-1,3 - DICHLOROPROPENE	2	0.00	0	0	0
t-1,3 - DICHLOROPROPENE	2	0.00	0	0	0
ACROLEIN	26	0.00	0	0	0

(continued)

TABLE 3-6. (Continued)

CHEMICAL NAME	No. of Times Quantified	Average Conc. ppm	Average Conc. Detected ppm	Highest Conc. ppm	Lowest Conc. ppm
1,4 -DICHLOROBENZENE	28	0.00	0	0	0
BROMOFORM	28	0.00	0	0	0
1,3 - DICHLOROPROPANE	26	0.00	0	0	0
1,2 - DICHLOROBENZENE	29	0.00	0	0	0
1,3 - DICHLORBENZENE	29	0.00	0	0	0
DIBROMOCHLOROMETHANE	28	0.00	0	0	0
BROMOMETHANE	28	0.00	0	0	0

[a]References 75-81.

landfills will be small initially, but with time, these landfills will become the major contributor to nationwide MSW landfill air emissions.

A summary of the estimated 1997 baseline emissions from each category of MSW landfills is presented in Table 3-7. As shown in this table, total NMOC emissions from MSW landfills are estimated to be 530,000 Mg per year in 1997. Of this total, existing landfills are expected to account for 510,000 Mg per year (98 percent of the total) and new landfills are expected to account for 9,300 Mg per year (2 percent of the total). Assuming that waste disposal volumes will remain about the same, the nationwide emissions from MSW landfills are expected to remain roughly constant. However, the contribution from each of the three landfill categories defined above is expected to change. The expected contribution of each MSW landfill category with respect to time is illustrated in Figure 3-4.

The baseline emission estimates presented in Table 3-7 were developed using three sources of information in combination with the Scholl Canyon gas generation model discussed in 3.2.4.7. These are: (1) results of the 1987 EPA MSW landfill survey, (2) the available data on gas generation rates, and (3) the available data on NMOC concentrations in landfill gas.

In 1986, EPA sent municipal landfill survey questionnaires to 1,250 of the estimated 6,034 active MSW landfills in the United States. From this survey, EPA received responses for a total of 1,174 active MSW landfills. Of these 1,174 landfills, the information provided on location (latitude and longitude), annual waste acceptance rate, refuse in place, age, depth, and design capacity were complete for 931 landfills. The landfill characteristics reported for these 931 landfills formed the basis for all national impacts presented in this document.

The EPA survey was designed to provide a stratified sample of both large and small municipal landfills and the design of the survey was considered in extrapolating from the 931 responses used up to the national total. Of the 931 landfill responses used, 151 were for large landfills and 780 were for small landfills. In comparison, EPA estimated that 362 of the 6,034 active municipal landfills were large and 5,672 were small when

TABLE 3-7. 1997 NATIONAL BASELINE EMISSION ESTIMATES

Landfill category	Number of landfills	Methane emissions (Mg/year)	NMOC emissions (Mg/year)
Existing MSW Landfills (Active and Closed)	7,480	1.8×10^7	510,000
New MSW Landfills	928	5.3×10^5	10,000
ALL AFFECTED LANDFILLS	8,408	1.8×10^7	520,000

designing the survey. Therefore, the following scale factors were developed for large and small landfill responses:

Large landfill scaling factor = 362/151 = 2.40
Small landfill scaling factor = 5,672/780 = 7.27

These scale factors were used to extrapolate the estimated baseline emissions from each of the 931 landfills up to the nationwide total.

The second source of information used to develop national baseline emission estimates was gas generation rate data. As discussed in Section 3.2.4.7, the gas generation rate is a function of time and the time dependent behavior can be predicted using models such as the Scholl Canyon model. The use of this model does, however, require two landfill specific constants (k and L_0), as well as the landfill characteristics. If sufficient gas generation data were available for a given landfill, the values of k and L_0 could be determined by regressing the measured gas generation rate versus time. Such data were not available for any landfills, but one time gas generation rate determinations were available for 54 landfills.[82] In the absence of time dependent data, values of k were back-calculated from the measured flow for a low, medium, and high value of L_0 using the Scholl Canyon model equation. Ultimate gas generation rate (L_0) values of 2,100, 6,350, and 8,120 ft^3/Mg (59.5, 179.8 and 230 m^3/Mg) of refuse were selected as high, medium, and low values, (or 80th, 50th and 20th percentile values) respectively, based on available information sources.[82] Using this approach, a total of 139 sets of k and L_0 were developed from the available gas generation data. In approximately 20 cases, a value for k could not be calculated for a given L_0 due to the lack of convergence on a single value. These sets of k and L_0, presented in Table 3-8, were randomly assigned to each of the 931 landfills.

The third source of information used in developing national baseline emission estimates was the available NMOC concentration data for landfill gas. Such data were available for landfill gas collected at 23 landfills. If there was more than one test result, the most recent data was used. If multiple results were provided, the arithmetic average was used.[82] These

TABLE 3-8. VALUES FOR k AND L_o

	k (1/yr)	L_o (ft^3/Mg)[a]
1	0.011	6,350
2	0.008	8,120
3	0.050	2,100
4	0.010	6,350
5	0.008	8,120
6	0.006	2,100
7	0.002	6,350
8	0.002	8,120
9	0.006	2,100
10	0.002	6,350
11	0.001	8,120
12	0.029	2,100
13	0.008	6,350
14	0.006	8,120
15	0.028	6,350
16	0.019	8,120
17	0.024	2,100
18	0.006	6,350
19	0.004	8,120
20	0.038	8,120
21	0.021	6,350
22	0.015	8,120
23	0.047	2,100
24	0.010	6,350
25	0.008	8,120
26	0.026	2,100
27	0.007	6,350
28	0.007	8,120
29	0.022	6,350
30	0.015	8,120
31	0.026	6,350
32	0.017	8,120
33	0.025	2,100
34	0.006	6,350
35	0.004	8,120
36	0.014	6,350
37	0.011	8,120
38	0.024	6,350
39	0.017	8,120
40	0.028	6,350
41	0.019	8,120
42	0.060	2,100
43	0.012	6,350
44	0.009	8,120
45	0.048	6,350
46	0.030	8,120

TABLE 3-8. (Continued)

	k (1/yr)	L_o (ft^3/Mg)[a]
47	0.020	6,350
48	0.015	8,120
49	0.049	6,350
50	0.033	8,120
51	0.031	6,350
52	0.024	8,120
53	0.140	6,350
54	0.080	8,120
55	0.041	2,100
56	0.006	6,350
57	0.005	8,120
58	0.009	2,100
59	0.002	6,350
60	0.002	8,120
61	0.016	6,350
62	0.011	8,120
63	0.015	6,350
64	0.012	8,120
65	0.075	2,100
66	0.012	6,350
67	0.009	8,120
68	0.150	2,100
69	0.017	6,350
70	0.012	8,120
71	0.085	2,100
72	0.015	6,350
73	0.011	8,120
74	0.046	2,100
75	0.011	6,350
76	0.008	8,120
77	0.030	6,350
78	0.022	8,120
79	0.070	2,100
80	0.015	6,350
81	0.011	8,120
82	0.026	6,350
83	0.019	8,120
84	0.130	2,100
85	0.019	6,350
86	0.014	8,120
87	0.011	2,100
88	0.003	6,350
89	0.003	8,120
90	0.021	2,100
91	0.006	6,350
92	0.005	8,120

TABLE 3-8. (Continued)

	k (1/yr)	L_o (ft^3/Mg)[a]
93	0.026	6,350
94	0.019	8,120
95	0.029	6,350
96	0.021	8,120
97	0.030	6,350
98	0.022	8,120
99	0.041	6,350
100	0.029	8,120
101	0.034	6,350
102	0.024	8,120
103	0.140	2,100
104	0.021	6,350
105	0.016	8,120
106	0.060	2,100
107	0.014	6,350
108	0.011	8,120
109	0.025	2,100
110	0.007	6,350
111	0.006	8,120
112	0.120	2,100
113	0.021	6,350
114	0.016	8,120
115	0.210	2,100
116	0.027	6,350
117	0.020	8,120
118	0.035	6,350
119	0.026	8,120
120	0.036	6,350
121	0.026	8,120
122	0.023	2,100
123	0.007	6,350
124	0.006	8,120
125	0.041	2,100
126	0.012	6,350
127	0.009	8,120
128	0.010	2,100
129	0.003	6,350
130	0.003	8,120
131	0.040	2,100
132	0.012	6,350
133	0.009	8,120
134	0.065	2,100
135	0.018	6,350
136	0.014	8,120
137	0.065	2,100
138	0.019	6,350
139	0.015	8,120

[a]To convert to m^3/Mg use the following conversion:

$$1 ft^3/Mg = .028\ m^3/Mg$$

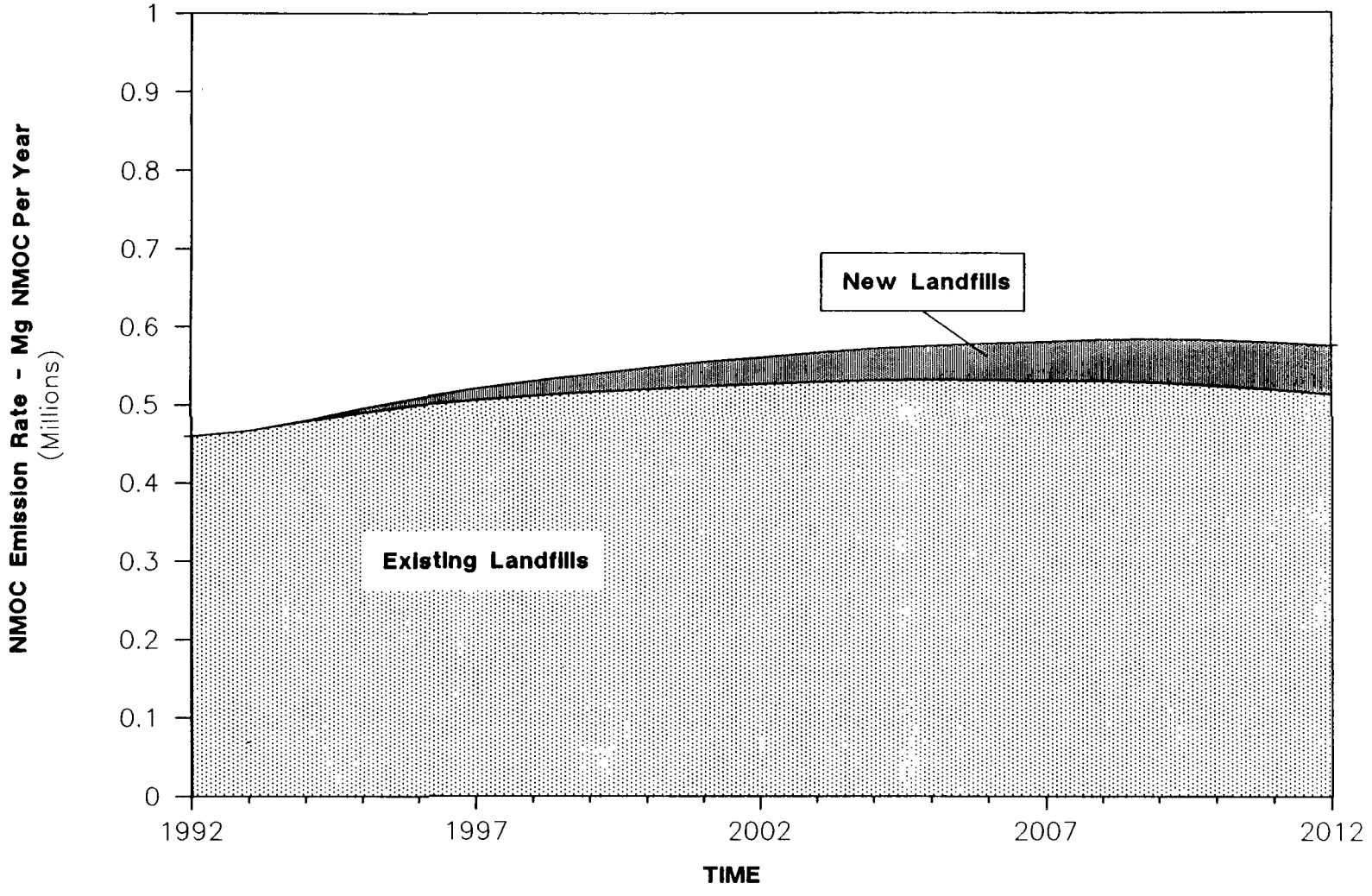


Figure 3-4. Time-dependent contribution to national baseline emissions.

23 NMOC concentrations, provided in Table 3-8, were randomly assigned to each landfill in the database.

The three information sources described above were used in combination with the Scholl Canyon model to develop baseline emission estimates for each category of municipal landfills. However, as indicated in the following subsections the approach varied slightly for each landfill category.

3.3.1 Baseline Emissions From Existing Active Landfills

As mentioned above, the EPA survey of MSW landfills was completed in 1987. Between the time the survey was conducted and the effective date of regulations being considered (expected to be 1992), many of the landfills included in the survey are expected to reach design capacity and close. In addition, it is expected that a number of landfills will be constructed and will begin to accept municipal refuse between 1987 and 1992. The location and size of these additional landfills is not know, but one would expect these newly opened landfills to be located near the landfills projected to close. It was also assumed that the newly opened landfills would closely resemble the landfills they replace in terms of physical and operating characteristics. This assumption was made for the sake of a qualitative analysis. In actuality, the newly opened landfills may be bigger and fewer in numbers. Based on this premise, EPA has projected the general location and characteristics of active landfills in 1992 using the 1987 survey data. For each of the 931 landfills active in 1987, the refuse in place has been projected in the year 1992 using information reported in the 1987 survey. If the landfill was projected to reach capacity before 1992, then a landfill with the same physical and operating characteristics has been projected to replace the closed landfill. Therefore, the overall number of landfills and national acceptance rate have been assumed to remain constant.

The baseline methane generation rate was estimated for each of the 931 landfills using the Scholl Canyon model discussed in Section 3.2.4.7, the projected landfill characteristics in 1992, and the assigned set of k and L_0. The methane generation rate was then multiplied by 2 to estimate the total gas generation rate (since methane accounts for only half of the landfill gas composition). This estimate of total landfill gas generation rate was then multiplied by the assigned NMOC concentration to estimate the

baseline NMOC emission rate. After estimating the baseline emission rate for each of the 931 landfills, the national baseline emission rate was estimated. The emission rate estimated for each landfill was multiplied by the appropriate scale factor and the scaled emission estimates were summed for all 931 landfills.

3.3.2 Baseline Emissions From Existing Closed Landfills

The baseline emissions from existing closed landfills were estimated very much the same as for existing active landfills. The only difference was the set of landfills and their characteristics. As mentioned above, existing closed landfills in this document are defined as those landfills which received municipal refuse prior to November 7, 1987, but reached capacity and closed prior to March 1, 1992. The location and characteristics of landfills in this category were determined from the 1987 EPA survey of active municipal landfills. The reported refuse in place, annual acceptance rates, and design capacities were used to project the landfills closing between 1987 and 1992. Based on the EPA municipal landfill survey 231 of the 931 landfills included in the impact analyses are expected to reach capacity and close by 1992. Applying the scaling factors presented above, these 231 landfills represent a total of 1,446 landfill nationwide.

The nationwide emission estimates for closed landfills were developed using the Scholl Canyon model presented in Section 3.2.4.7, assigned sets of k and L_0, assigned NMOC concentrations, and the appropriate scale factors. Emission estimates for methane and NMOC were developed for each landfill, then scaled and summed to estimate total nationwide emissions.

3.3.3. Baseline Emissions From New Landfills

The physical and operating characteristics of new landfills were projected based on the EPA survey of active landfills in 1987 and the premise that new landfills will have characteristics similar to the landfills they replace. Information on refuse in place, annual acceptance rates, and design capacities provided in the EPA survey were used to project landfills reaching capacity and closing between 1992 and 1997. For each landfill projected to close during this time period, a new landfill with identical physical and operating characteristics was assumed to open. with

a few exceptions. If the landfill that closes is a co-disposal site (i.e., had been assigned an NMOC concentration obtained from a co-disposal landfill), the new landfill was randomly re-assigned a non-codisposal NMOC concentration. As a result of RCRA regulations, co-disposal practices are not expected to new landfills. If the new landfill was projected to close in less than 20 years, the design capacity was modified so that the landfill would be active for at least 20 years at the same rate of waste acceptance. If the new land fill was projected to stay active in excess of 200 years, the design capacity was modified to yield a maximum active life of 200 years at the same acceptance rate. Using this approach, a total 143 new landfills were projected to open, from the set of 931 landfills. Applying the appropriate scale factors, these 143 landfills represent 928 landfills nationwide.

Emissions from the projected 928 new landfills were also estimated using the Scholl Canyon model, assigned values of k and L_0, assigned values of NMOC concentration, and the appropriate scale factors. However, one difference should be noted. Emission estimates for methane and NMOC were developed for each projected landfill, then scaled and summed to estimate total nationwide emissions.

3.3.4 State Regulations

In the past, the regulation of emissions from MSW landfills has mostly been associated with controlling methane migration/explosion potential and odor nuisance under RCRA. Within the last several years, however, a small number of state and local jurisdictions have commissioned special studies to assess the potential human health and environmental impacts associated with landfill air emissions. Table 3-9 summarizes the state regulations that address the control of air emissions from municipal solid waste landfills.

As summarized in Table 3-9, 27 states have implemented laws regulating air emissions from municipal solid waste landfills. California is the only state, at the present time, that has implemented air emissions regulations for landfills under the state's air pollution control authority. The other states have implemented landfill air emissions regulations under solid waste laws.

TABLE 3-9. SUMMARY OF STATE REGULATIONS CONTROLLING AIR EMISSIONS FROM MSW LANDFILLS

State	Control criteria	Collection/control system requirements	Testing/reporting requirements	Exemption criteria	Reference number
Alaska	conc. of gases >LEL[a]	some form of venting, or other controls	None	None	83
California	levels of tested air contaminants pose a health risk avg. max. conc. of total organics over a certain area >500 ppm max. conc. of organic compounds as methane at any >500 ppm	flaring, internal combustion engine, or gas treatment and sale	chemical characterization of gas on and off-site monitoring probes at landfills perimeter to detect gas migration periodic sampling and testing of methane and toxics, and testing of the efficiency of of controls	in-place (RIP) tonnage <1,000,000 tons	84, 85
Delaware	conc. of gases >LEL[a]	venting	if monitoring required, then it must remain in place at least 5 years, quarterly gas composition data must be taken, and quarterly gas generation rate data may be required	None	86
Florida	to prevent explosion and fires, damage to vegetation, and objectional odors off-site	site specific design requirements, and must prevent lateral movement of gases by collection	None	None	87
Illinois	prevention of air pollution	None	None	None	88
Indiana	methane conc. >25% of the LEL[a] within facility structures or >LEL[a] at the facility boundary	None	a methane monitoring program approved by the commissioner must implemented	None	89
Kansas	methane conc. >25% of the LEL[a] within facility structures or at the facility boundary	None	None	None	90

(continued)

TABLE 3-9. (Continued)

State	Control criteria	Collection/control system requirements	Testing/reporting requirements	Exemption criteria	Reference number
Kentucky	methane conc. >25% of the LELs within facility structures or >LEL[a] at the facility boundary sites within 500 ft of a residential, farm, commercial or industrial building submit a methane gas contingency control plan	None	None	None	91
Louisiana	cells containing material and meeting criteria of LAC 33:VII. 1305.D.7.a.ii must be connected to a gas control system	venting or gas dispersal into the air	monthly surveys must be conducted, upon request of the department, for the presence of strong odors	None	92
Maine	methane conc. >25% LEL[a] within facility structures or >LEL[a] at the facility boundary	None	None	None	93
Maryland	methane conc. >25% of LEL[a] within facility structures or >LEL[a] at the facility boundary	None	None	None	94
Michigan	if gases present a a hazard to those operating the fill or living and working nearby	a means of assuring that gases cannot travel laterally or accumulate in structures must be designed and employed	None	None	95
Minnesota	if gases are found to migrate laterally, or explosive conc. reached	venting, or other means approved by the commissioner	None	None	96

(continued)

TABLE 3-9. (Continued)

State	Control criteria	Collection/control system requirements	Testing/reporting requirements	Exemption criteria	Reference number
Mississippi	if the future use of a landfill involves a recreational park	None	control installation delayed until significant gas releases have been detected or until closure procedures are initiated	None	97
Missouri	methane conc. >25% of LEL[a] within facility structures or >5% of the LEL[a] at the facility boundary	venting or flaring	those facilities required to monitor gases must submit the results to the department	None	98
New Hampshire	methane conc. >25% LEL[a] within facility structures or >50% of the LEL[a] at the facility boundary	None	None	None	99
New Jersey[b]	if methane is found to accumulate in any structure, causing a potential hazard if potential damage to vegetation beyond the perimeter is present methane conc. >25% of the LEL[a] within facility structures or at the facility boundary	venting, collection, or combustion	gas samples must be taken before, and after combustion, and methane gas sensors must be installed to trigger an alarm when methane gas is detected	None	100
New York	methane conc. >25% of the LEL[a] within facility structures or >LEL[a] at the facility boundary	None	None	None	101
North Dakota	if lateral migration occurs, creating a potentially hazardous condition	venting, or other means approved by department	None	None	102

(continued)

TABLE 3-9. (Continued)

State	Control criteria	Collection/control system requirements	Testing/reporting requirements	Exemption criteria	Reference number
Oregon	methane conc. >25% of the LEL[a] within facility structures >LEL[a] at the facility boundary odor becomes a becomes a public nuisance	None	the department may require gas samples to be taken at a specified interval and submit the results of an analysis within a specified time frame	None	103
Pennsylvania	all sites must install controls	venting	gas monitoring must be installed	None	104
Rhode Island	if lateral movement gases or accumulation of gases in confined structures occurs	venting	None	None	105
South Dakota	if department considers necessary	None	None	None	106
Texas	control of air pollution	venting	None	None	107
Virginia	gases must be controlled	None	a gas management plan and gas monitoring procedures must be implemented	None	108
Washington	methane conc. >25% of the LEL[a] within facility structures or >LEL[a] at the facility boundary conc. of gases >100 ppmv of hydrocarbons in off-site structures	collection and sale flaring utilized for energy value	None	acceptance rate <10,000 cubic yards/year or little or no gases will be produced	109

(continued)

TABLE 3-9. (Continued)

State	Control criteria	Collection/control system requirements	Testing/reporting requirements	Exemption criteria	Reference number
Wisconsin	methane conc. >25% of the LEL[a] within facility structures or > the lower detection limit at the facility boundary must collect and combust all hazardous air contaminants	venting	gas monitoring probes must be installed outside the limits of the landfill	None	110
Wyoming	violation of Air Quality Regulations	None	None	None	111

[a]LEL (lower explosive limit) means the lowest percent by volume of a mixture of gas which will propagate a flame in air at 25°C atmospheric pressure.

[b]New Jersey's Solid and Hazardous Waste Management Regulations provide extensive design and sampling specifications for a landfill gas collection system.

Twelve of the 28 states regulate air emissions from landfills based on the methane concentration in or near the landfill. Five states base control criteria on the potential for lateral migration of the landfill gas, which could result in off-site hazards (such as explosions and fires) and/or odor nuisances. Four states have regulations that simply state that landfills must control air pollution. California and Washington base control criteria on the levels of tested air contaminants, while Louisiana bases control on the properties of the material in the landfill. Pennsylvania is the only state that requires all landfills to install controls, regardless of gas concentration or the type of waste deposited.

Uncontrolled venting was found to be a generally accepted method for controlling the emissions from landfills, while several states also encourage flaring, internal combustion, and treatment and sale of the gas. Twelve states mention no specific requirement for the type of controls that must be installed.

3.4 EXPLOSION HAZARDS AND ODOR NUISANCE

3.4.1 Explosion Hazards

Methane, a major component of landfill gas, is highly explosive when it is present in air at a concentration between 5.5 and 15 volume percent. The concentration of methane produced during the bacterial decomposition of municipal wastes typically exceeds the upper explosion limit. However, as methane migrates outside of the landfill perimeter, it can be diluted by air to explosive concentrations.

Landfill gas migration occurs because of the pressure gradient developed by landfill gas generation through the biodegradation of refuse. The landfill gas moves toward low pressure areas through pathways of least resistance. The extent to which landfill gas migrates laterally instead of vertically depends on where the pathways of least resistance are located. If the landfill surface layers are relatively impermeable, there will be a greater tendency for gas to migrate laterally out of the landfill. Natural and man-made barriers can reduce the permeability of the landfill surface layers. Such barriers include clay deposits, a high water table, compacted subgrade, and wet or frozen surface soil. Lateral gas migration can also be

enhanced if adjacent soils are relatively permeable or corridors for gas movement exist. Some examples of gas corridors include storm sewer culverts, and buried utility lines. Landfill gases have reportedly migrated as far as 300 meters into structures located on or near the landfill. In addition to the danger of explosion, migrating landfill gas can also displace air in enclosed areas and cause asphyxiation of individuals in these areas.[112]

The U. S. Environmental Protection Agency has promulgated regulations for controlling explosive gases from sanitary landfills based on the methane concentrations in structures built on the landfill and in the soil at the property boundary.[113] The rule states that the concentration of explosive gases generated by a facility should not exceed 1.25 volume percent methane (25 percent of lower explosion limit) in facility structures and 5.5 volume percent methane at the property boundary. The revised Subtitle D Criteria, proposed 8/88, requires monitoring of the LEL facility structures and the property boundary.

3.4.2 Odor Nuisance

Landfill gas has a distinctive odor due to trace vapors which are present at low concentrations in the gas. This odor is generally regarded as unpleasant, and it can cause a considerable environmental nuisance in the vicinity of the site. A transport of odors from the landfill to neighboring sites is affected by such factors as the rate of gas production, operating practices (refuse coverage depth and materials used), and the local topography.

Odorous compounds in landfill gas are formed during the refuse decomposition process. The presence of significant quantities of industrial wastes or household solvents can increase the number of compounds released. The major contribution to odor comes from two groups of compounds. The first group is dominated by esters and organosulfurs, but also includes butan-2-ol and certain solvents which may have been deposited with the waste. These compounds are not widespread and are variable in their concentrations. The second group is widespread and includes alkyl benzenes and limonene. Together, with other hydrocarbons, these are probably responsible for the background smell associated with a landfill. Typical

TABLE 3-10. HIGHLY ODOROUS COMPONENTS OF LANDFILL GAS

Compound	Odor threshold (mg/m^3)
Group A	
Limonene	0.06
Xylenes	0.4
Ethyl benzene	0.2
Propyl benzenes	0.04
Butyl benzenes	0.1
Group B	
Methanethiol	4×10^{-5}
Dimethyl sulfide	0.01
Butan-2-ol	0.1
Methyl butanoate	0.005
Ethyl propionate	0.1
Ethyl butanoate	0.003
Propyl propionate	0.1
Butyl acetate	0.03
Propyl butanoate	0.1
Dipropyl ethers	0.07

[a]Reference 114.

odorous compounds which may be present in the landfill gas are listed in Table 3-10. Included in this table are the odor thresholds for each compound.[114]

The majority of malodorous compounds are formed during the anaerobic nonmethanogenic and anaerobic stages of decomposition. During the early stages of decomposition, alcohols are particularly noticeable. Initial ethanol concentrations may exceed 1 g/m^3. The sweet, putrid smells of these compounds lead to the most penetrating pulmes which become less potent with time. The orhanosulfurs are also well represented in the early stages of decomposition. These usually overpower the hdyrogen sulfude which is typically present at concentrations between 0.1 and 20 mg/m^3.[115,116] No major odor problems should be associated with the final stage of decomposition the anaerobic methanogenic as the gases formed are not themselves odorous. However, the presence of methane has been reported to enhance perception of other malodorants.[117]

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4. LANDFILL GAS COLLECTION AND CONTROL TECHNIQUES

Control of municipal landfill air emissions requires both effective collection of the generated landfill gases and effective recovery or destruction of organics in the collected gas. This chapter describes the gas collection and control technologies that can be used to control methane and nonmethane organic compound (NMOC) emissions from municipal landfills. The effective design of gas collection systems is discussed in Section 4.1, and applicable control devices are discussed in Section 4.2. The advantages and disadvantages of the various control techniques, in terms of environmental impacts, are compared in Section 4.3.

4.1 LANDFILL GAS COLLECTION TECHNIQUES

Landfill collection systems can be categorized into two basic types: active collection systems and passive collection systems. Active collection systems employ mechanical blowers or compressors to provide a pressure gradient in order to extract the landfill gas. Passive collection systems rely on the natural pressure gradient (i.e., internal landfill pressure created due to landfill gas generation) or concentration gradient to convey the landfill gas to the atmosphere or a control system.

Based on theoretical evaluations, well-designed active collection systems are considered the most effective means of gas collection.[1] Generally, passive collection systems have much lower collection efficiency since they rely on natural pressure or concentration gradient as a driving force for gas flow rather than a stronger, mechanically-induced pressure gradient. A passive system, however, can be nearly equivalent in collection efficiency to an active system if the landfill design includes synthetic liners on the top, bottom, and sides of the landfill.

Active collection systems can be further categorized into two types: vertical well systems and horizontal trench systems. Vertical well systems are discussed in Section 4.1.1. Passive systems are discussed in Section 4.1.2. The type of collection system employed often depends on the landfill characteristics and landfill operating practices. For example, if a landfill employs a layer-by-layer landfilling method (as compared to

cell-by-cell methods), an active horizontal trench collection system may be preferred over an active vertical well collection system due to the ease of collection system installation. However, if the water table extends into the refuse, a horizontal trench system has a tendency to flood, thus decreasing the collecting efficiency. Applications, advantages, and disadvantages of different collection systems are summarized in Table 4-1.

4.1.1 Active Collection Systems

Active collection systems employ mechanical blowers or compressors to create a pressure gradient and extract the landfill gas. A typical active collection system with extraction wells is shown in Figure 4-1. Active collection systems consist of two major components:

- Gas extraction wells and/or trenches, and
- Gas moving equipment (e.g., piping and blowers)

4.1.1.1 Gas Extraction Wells/Trenches. Gas extraction wells may be installed in the landfill refuse or along the perimeter of the landfill. For a landfill that is actively accepting waste, wells are generally installed in the capped sections. Additional wells are installed as more refuse is accumulated.

The wells consist of a drilled excavation 12 to 36 inches in diameter. A 2 to 6 inch diameter pipe (PVC, HDPE, or galvanized iron) is placed in the well, and the well is filled with 1-inch diameter or larger, crushed stone. The pipe is perforated in the area where gas is to be collected but solid near the surface to prevent air infiltration. A typical extraction well is shown in Figure 4-2.

In unlined landfills, gas extraction wells are usually drilled to the depth of the ground water table or to the base of the landfill, whichever is less. In lined landfills, wells are typically drilled to only 75 percent of the landfill depth to avoid damaging the liner system. Typical well depths range from 20 to 50 feet but may exceed 100 feet. The spacing between gas extraction wells depends on the landfill characteristics (e.g., type of waste, degree of waste compaction, landfill gas generation rate, etc.) and the magnitude of pressure gradient applied by the blower or compressor. Typical well spacing ranges from 50 to 300 feet.

TABLE 4-1. COMPARISON OF VARIOUS COLLECTION SYSTEMS

Collection system type	Preferred applications	Advantages	Disadvantages
Active vertical well collection systems	Landfills employing cell-by-cell landfilling methods	Cheaper or equivalent in costs when compared to horizontal trench systems	Difficult to install and operate on the active face of the landfill (may have to replace wells destroyed by heavy operative equipment)
Horizontal trench collection systems	Landfills employing layer-by-layer landfilling methods Landfills with natural depressions such as canyon	Easy to install since drilling is not required Convenient to install and operate on the active face of the landfill	The bottom trench layer has higher tendency to collapse and difficult to repair once it collapses Has tendency to flood easily if water table is high Difficult to maintain uniform vacuum along the length (or width) of the landfill
Passive collection systems	Landfills with good containment (side liners and cap)	Cheaper to install and maintain if only a few wells are required	Collection efficiency is generally much lower than active collection systems Costs is generally higher than active systems when designed for the same collection efficiency

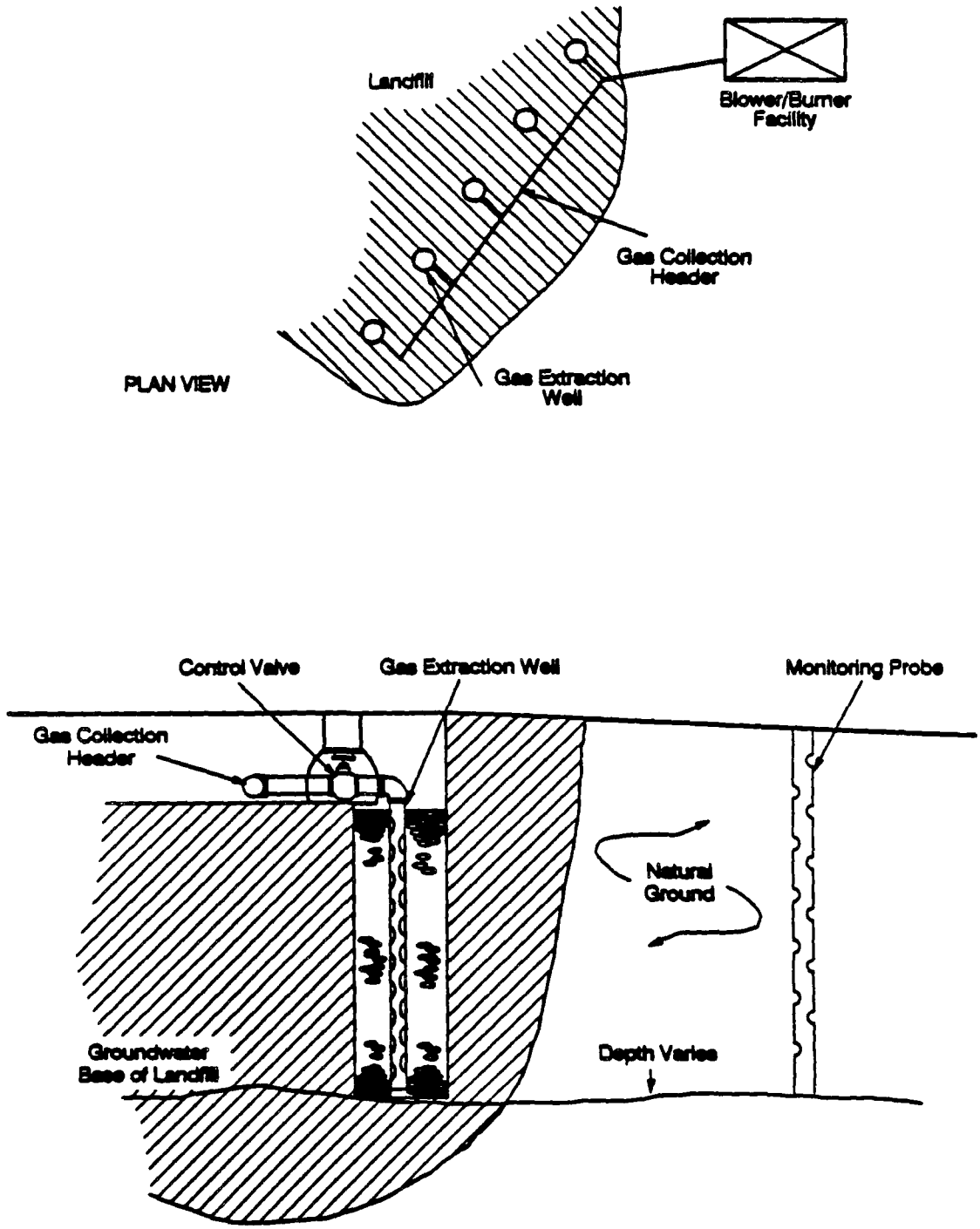
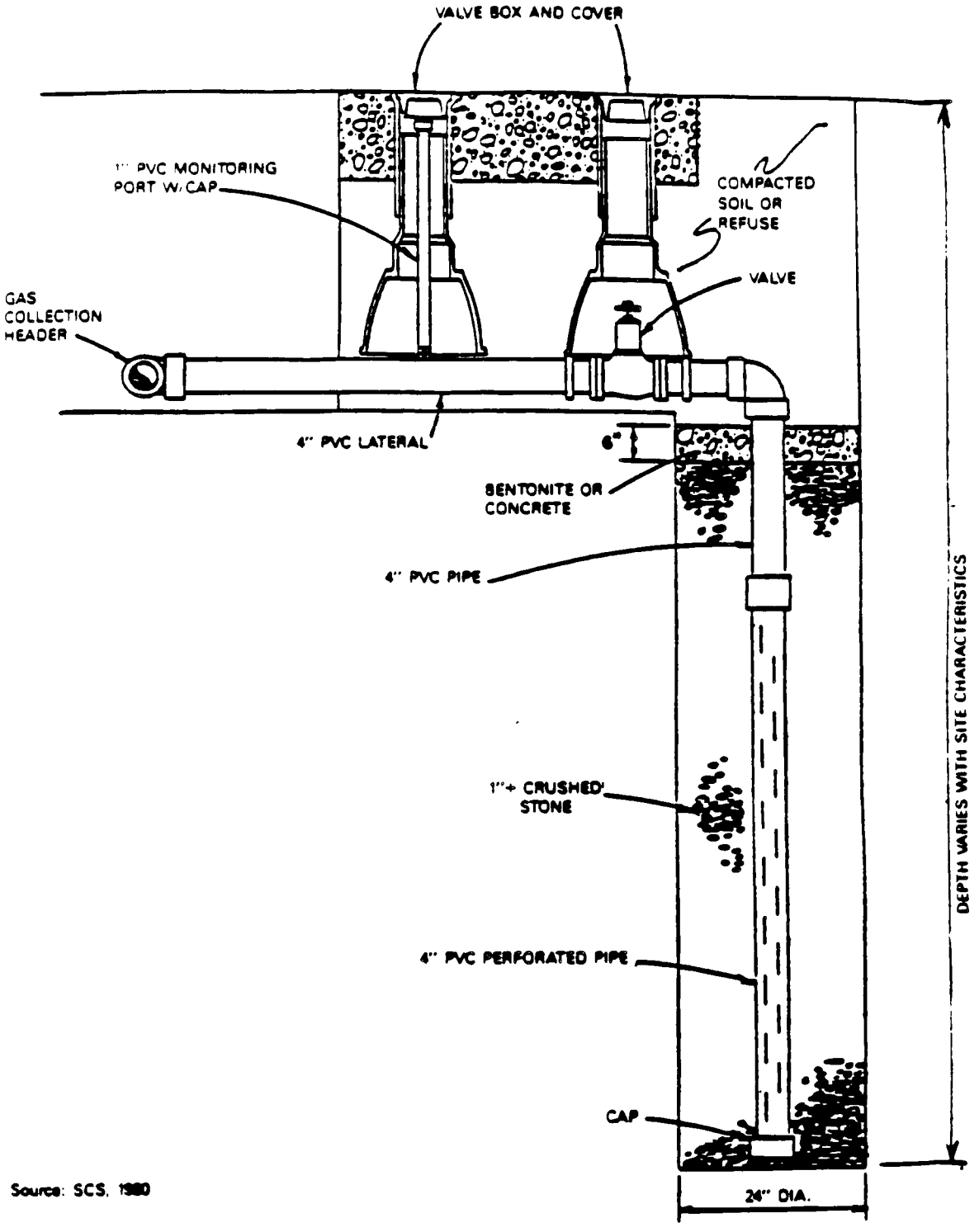


Figure 4-1. Active collection system.



Source: SCS, 1980

Figure 4-2. Extraction well.

Trenches may be installed instead of or in combination with wells to collect the landfill gas. The trenches can be vertical or horizontal at or near the base of the landfill. A vertical trench is illustrated in Figure 4-3. A vertical trench is constructed in much the same manner as a vertical well is constructed, except that it extends to the surface along one dimension of the landfill. Horizontal trenches are installed within a landfill cell as each layer of waste is applied. This allows for gas collection as soon as possible after gas generation begins and avoids the need for above-ground piping which can interfere with landfill maintenance equipment. A horizontal trench is illustrated in Figure 4-4.

4.1.1.2. Gas Moving Equipment. A gas collection header system conveys the flow of collected landfill gas from the well or trench to the blower/compressor facility. A typical header pipe is made of PVC or polyethylene and is 6 to 24 inches in diameter.

The collected landfill gas is conveyed through the header system by a blower or compressor. The size and type of compressor or blower depend on total gas flow rate, total system pressure drop, and vacuum requirements. For systems requiring only a small vacuum (up to 40 inches of water), centrifugal blowers are often used. Centrifugal blowers offer the advantage of easy throttling throughout their operation range. These blowers can accommodate total system pressure drops of up to 50 inches of water and can transport high flow rates (100 to 100,000 cfm). For lower flow rates and higher pressures, regenerative (combination of axial and centrifugal) blowers are often used.[2]

Rotary lobe or screw-and-piston type compressors are used when the system vacuum requirement is greater than 2 to 3 psi (55 to 85 inches of water) and high discharge pressures (>100 psig) are required for pipeline transport or processing. Systems with compressors have limited flow-throttling capability. Compressors are positive displacement type devices and excessive throttling of flow can damage the compressor.

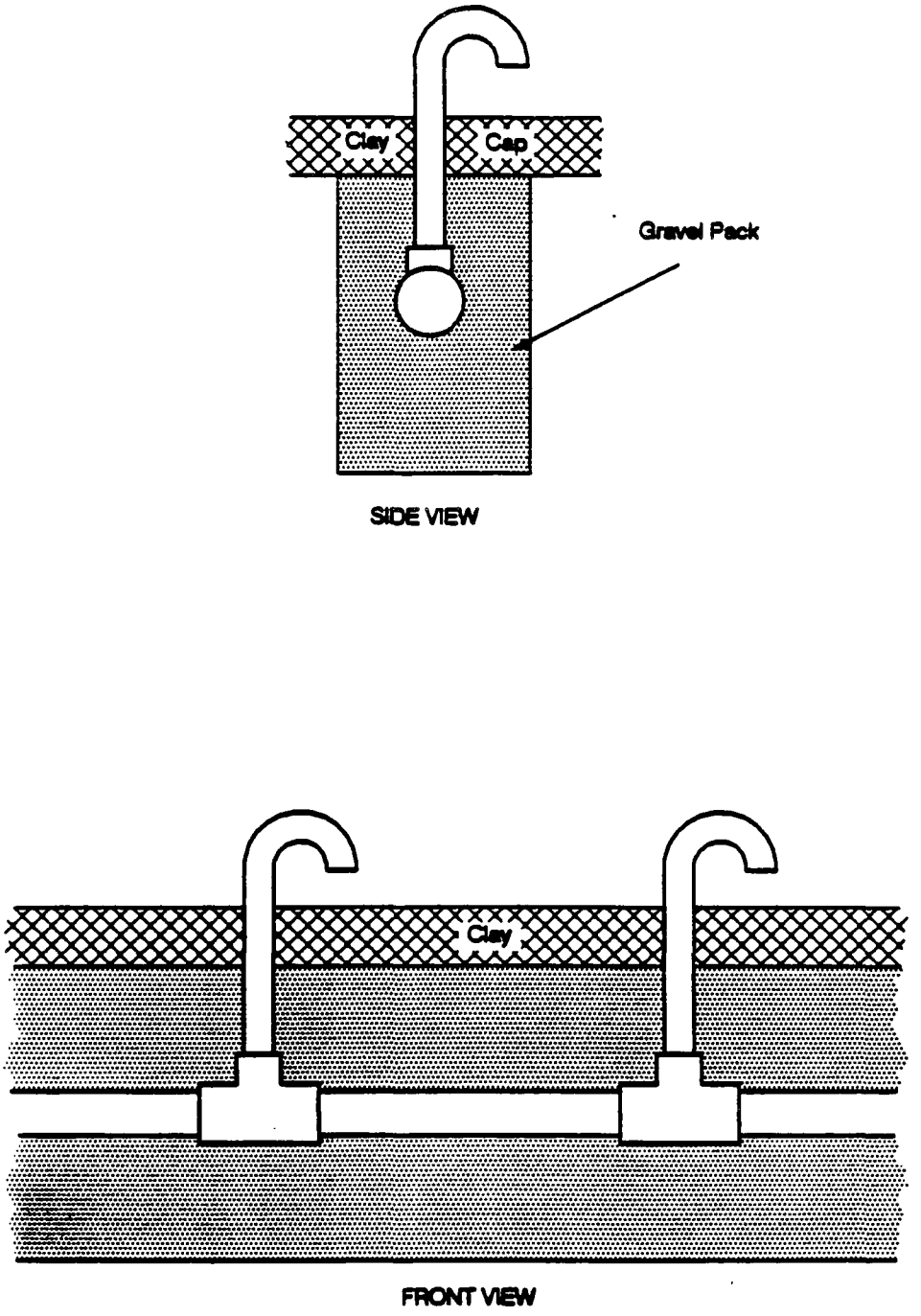


Figure 4-3. Vertical trench for an active collection system.

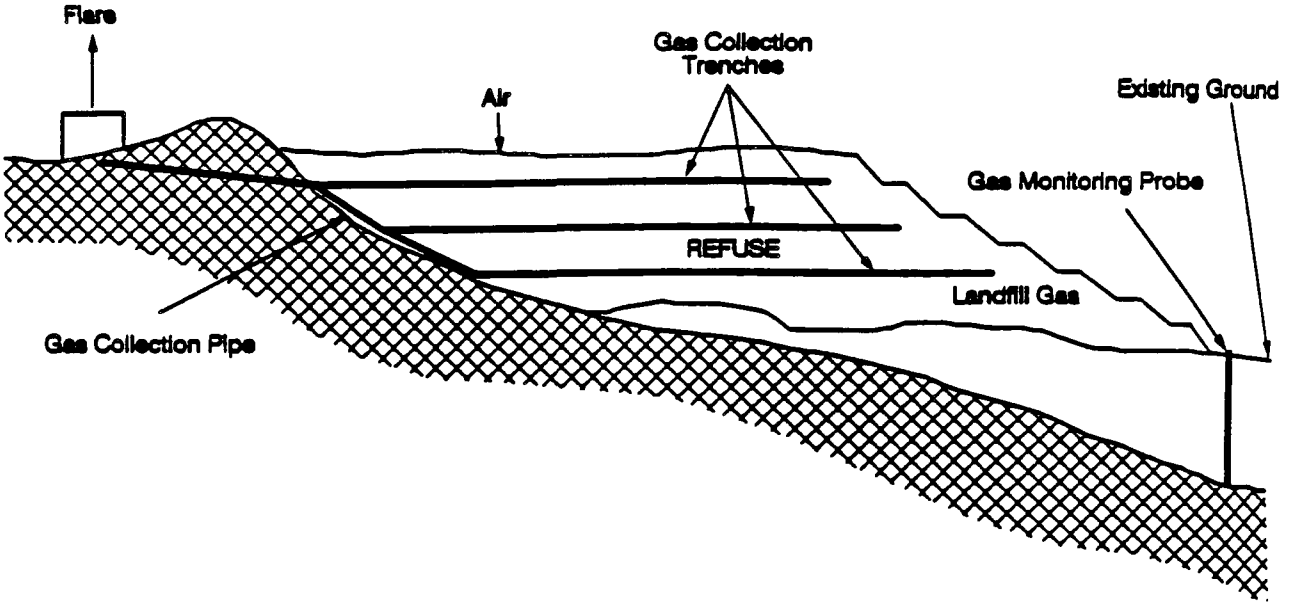


Figure 4-4. Horizontal trench collection system.

4.1.2 Passive Collection Systems

As indicated above, passive collection systems rely solely on natural pressure or concentration gradient in the landfill to capture the landfill gas. Like active collection systems, passive collection systems use extraction wells to collect landfill gas. The construction of passive collection wells is similar to that of active wells which is illustrated in Figure 4-2.

The well construction for passive systems is much less critical than active systems. This is primarily because the collection well is under positive pressure and air infiltration is not a concern. Additionally, elaborate well head assemblies are not required since monitoring and adjustment is not necessary. However, it is important that a good seal be provided around the passive well when synthetic cover liners are used. Either a boot type seal, flange type seal, concrete mooring or other sealing technique is typically used at each well location to maintain the integrity of the synthetic liner.

4.1.3 Effectiveness of Landfill Gas Collection

The purpose of this section is to provide some general criteria for evaluating the effectiveness of landfill gas collection systems. The collection efficiency has not been determined at any landfill. However, one landfill facility operator estimated that a well-designed system can typically collect about 50 to 60 percent of the gas generated within a landfill.[3,4]

The effectiveness of an active landfill gas collection system depends greatly on the design and operation of the system. From the perspective of air emission control, an effective active collection system design would include the following attributes:

- Gas moving equipment capable of handling the maximum landfill gas generation rate.
- Collection wells and trenches configured such that landfill gas is effectively collected from all areas of the landfill.
- Design provisions for monitoring and adjusting the operation of individual extraction wells and trenches.

An effective passive landfill gas collection system would also include a collection well or trench configuration that effectively collects landfill gas from all areas of the landfill. The efficiency of a passive collection system would also greatly depend on good containment of the landfill gas. An example of good containment would be synthetic liners on the top, sides and bottom of the landfill.

The first criteria that should be satisfied for an active system is gas moving equipment capable of handling the maximum landfill gas generation rate. Blowers or compressors and header pipes need to be sized to handle the maximum landfill gas generation rate. In addition, collection header pipes should also be sized to minimize pressure drop. The maximum landfill gas generation rate is highly variable but may be estimated using the range reported in one EPA study (0.001-0.0008 m^3 landfill gas/kg of dry refuse/yr).[5]

Each extraction well or trench has a zone of influence within which landfill gas can be effectively collected. The zone of influence of an extraction well or trench is defined as the distance from the well center to a point in the landfill where the pressure gradient applied by the blower or compressor approaches zero. The zone of influence determines the spacing between extraction wells or location of trenches since an effective collection system covers the entire area of the landfill. The zones (or radii) of influence for gas extraction wells are illustrated in Figure 4-5.

The spacing between extraction wells depends on the depth of the landfill, the magnitude of the pressure gradient applied by the blower or compressor, type of waste, degree of compaction of waste, and moisture content of gas. For perimeter extraction wells, additional variables such as the outside soil type, permeability of the soil, moisture content of the soil, and stratigraphy should be considered. One EPA study reports a typical well spacing to be 260 feet with a radius of influence of 150 feet.[6] These distances are based on a well extraction rate of 50 ft^3/minute and a well vacuum of 3 inches of water.

The desired method for determining effective well spacing at a specific landfill is the use of field measurement data. The EPA Method 2E can be used to determine the average stabilized radius of influence for both

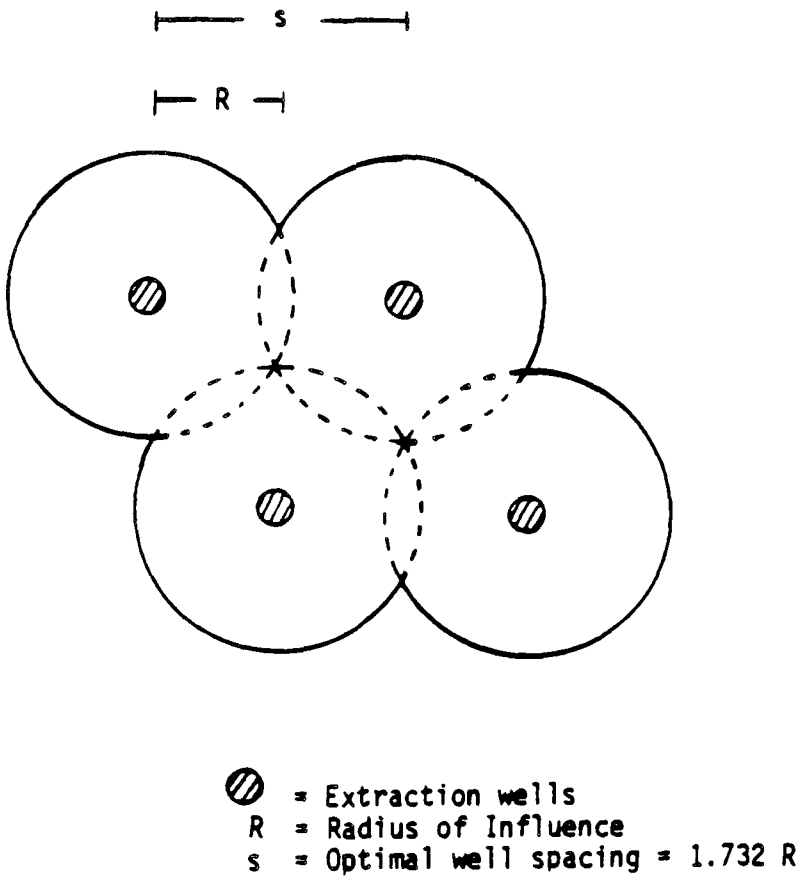


Figure 4-5. Zones of influence for gas extraction wells.

perimeter wells and interior wells. This measured radius of influence can then be used to site wells. A good practice is to place wells along the perimeter of the landfill (but still in the refuse) no more than the perimeter radius of influence from the perimeter and no more than two times the perimeter radius of influence apart. As illustrated in Figure 4-6, a helpful technique is to site the location of each well and draw a circle with radius equal to the radius of influence (perimeter radius of influence for perimeter wells and interior radius of influence for interior wells). Once the perimeter wells are sited on the landfill plot plan, the interior wells are sited at no more than two times the interior radius of influence in an orientation such that essentially all areas of the landfill are covered by the radii of influence.

In situations where field testing is not performed, the well spacing can be determined based on theoretical concepts. Understanding the behavior of landfill gas through the municipal landfill refuse and cover (final or daily cover) material is important in order to design the landfill gas collection system properly. The flow of landfill gas can be described by Darcy's Law. Darcy's Law correlates the flow of gas through a porous media as a function of the gas properties (e.g., density and viscosity), the properties of the porous media (e.g., permeabilities of refuse and cover), and pressure gradient.

When active collection systems (both vertical and horizontal) are designed, it is also important to understand the relationship between the magnitude of vacuum applied and the degree of air infiltration into the landfill. Excessive air infiltration into the landfill can kill the methanogens, which produce landfill gas from the municipal refuse. If excessive air infiltration continues, decomposition becomes aerobic and the internal landfill temperature can increase and possibly lead to a landfill fire. If the landfill conditions are such that air infiltration into the landfill is significant (e.g. highly permeable cover and/or shallow landfill), the magnitude of vacuum applied may need to be reduced to minimize the amount of air infiltration. Direct consequence of the reduced vacuum is an increased number of wells or trenches required to achieve the same collection efficiency. Therefore, consideration of air infiltration is

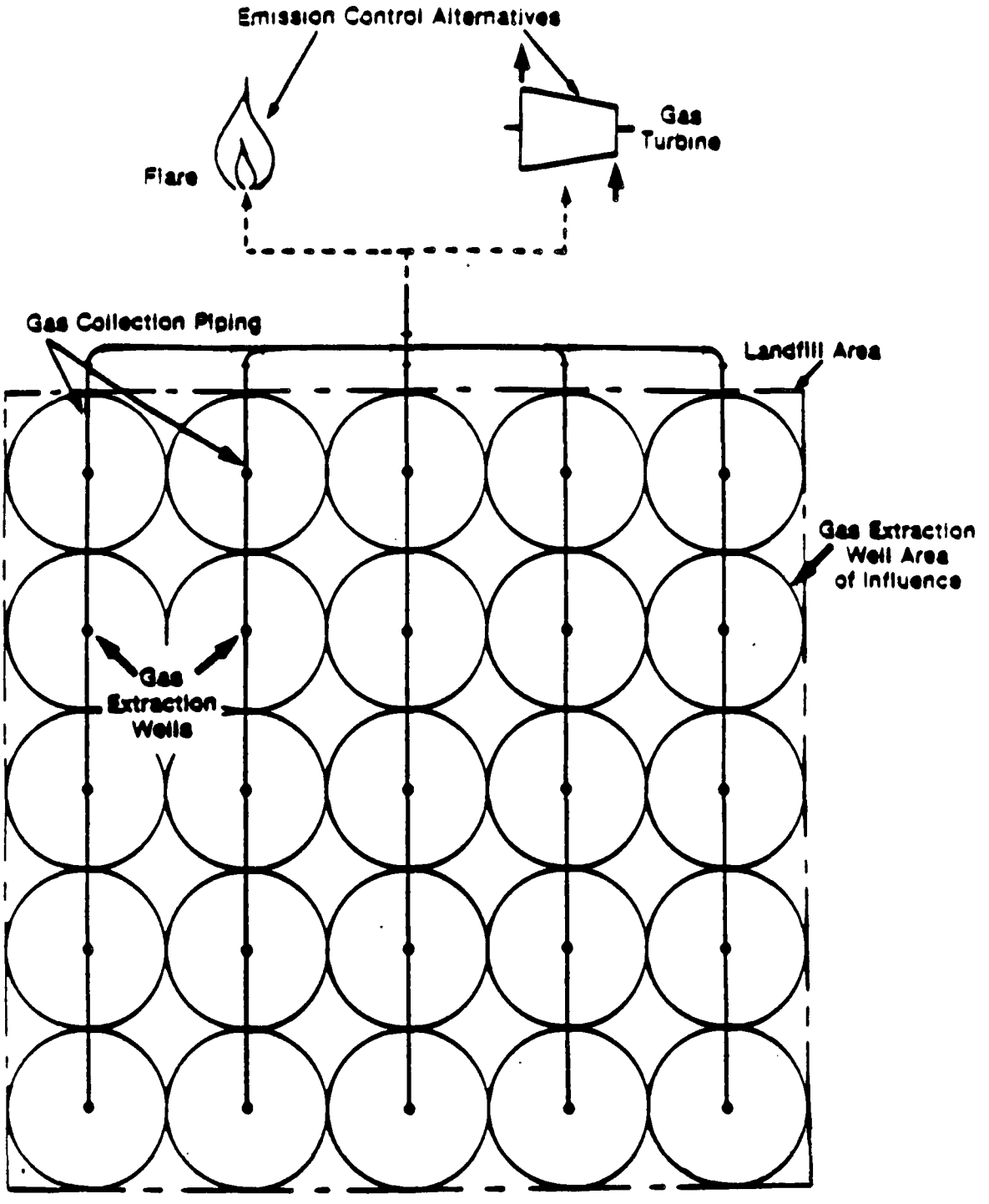


Figure 4-6. Technique for siting wells.

required in designing the active collection systems for shallow landfills. The problem of air infiltration does not exist for passive systems since passive systems rely on the natural pressure gradient (i.e., difference between atmospheric pressure and internal landfill pressure) rather than applying vacuum.

The theoretical approach, which can be used to design different types of landfill gas collection systems (active well systems, active horizontal trench systems, and passive systems), is based on specific landfill information. Information on the following landfill characteristics are used in the design equations:

- Landfill design capacity
- Average annual refuse acceptance rate
- Age of landfill upon closure
- Landfill depth
- Refuse methane generation potential, L_0
- Landfill gas generation rate constant, k
- Refuse permeability, k_{refuse}
- Cover permeability, k_{cover}

The first four parameters are usually readily available for a given landfill. The refuse methane generation potential (L_0) and the landfill gas generation rate constant (k) are the required inputs to the first order landfill gas generation rate model which is described in detail in Chapter 3 and they vary depending on the landfill characteristics such as the refuse composition, refuse moisture content, pH, and temperature. The values of L_0 and k must be assumed unless landfill specific test data are available. The values of k_{refuse} and k_{cover} also vary from landfill to landfill but can be estimated from the available literature values. Available literature values and actual data for L_0 and k may be found in a memorandum titled "Use of a Landfill Gas Generation Model to Estimate VOC Emissions from Landfills".[7] A detailed discussion of the theorectical approach for designing active vertical, active horizontal, and passive vertical collection systems is provided in Appendix H.

In a good design each extraction well or trench is equipped with a throttling valve and pressure gauge in order to adjust and monitor the collection system. In addition, the gas collection header system is designed so that water condensate can be separated from the collected gas (e.g., via sloping of the pipings or water traps at low points. Wells are also equipped with at least one sample port that can be used to monitor pressure and to collect gas samples periodically.

4.2 LANDFILL GAS EMISSION CONTROL/TREATMENT TECHNIQUES

There are two basic types of landfill gas control/treatment options available: (1) combustion of the landfill gas, and (2) purification of the landfill gas. The combustion techniques can further be categorized into two types: (1) combustion techniques which destroy organics without energy recovery, and (2) combustion techniques which recover energy from the destruction of organics.

The combustion techniques which do not recover energy are flares and afterburners. The energy recovery techniques include gas turbines, internal combustion engines, and boiler-to-steam turbine systems, all of which generate electricity from the combustion of landfill gas. Boilers may also be used at the landfill site or off-site to recover energy from landfill gas in the form of steam.

Purification techniques (adsorption, absorption, membranes) process raw landfill gas to pipeline quality natural gas. All purification techniques involve removal of water before removing carbon dioxide. The water is removed by either absorption with glycols or adsorption with silica gel, alumina, or molecular sieve. The removal method of nonmethane hydrocarbons depends on the different CO_2 removal techniques chosen and the composition of the landfill gas. Usually the same techniques used for CO_2 removal are also used to remove nonmethane hydrocarbons by simply adding an extra absorption, adsorption, or condensation step. Removal of nonmethane hydrocarbons is often an important part of the purification scheme. Standard natural gas pipelines generally do not accept halogenated compounds and sulfur derivatives. Consequently the removal of these compounds is also a significant part of process design.

The selection of a recovery technique versus a control technique is highly dependent on such factors as the landfill gas generation rate, the availability of a market for the recovered energy, and environmental impacts. If the landfill characteristics are such that the landfill does not produce enough gas to economically support combustion techniques with energy recovery (i.e., gas turbines, internal combustion engines, boiler/steam turbines) or purification techniques, flaring may be best suited for the specific landfill. Developers of landfill gas recovery systems cite the following factors as necessary for economically feasible landfill gas recovery projects: (1) refuse in place of greater than 2 million tons (1.8 million Mg), (2) depth of refuse greater than 35 feet, (3) landfill area of greater than 35 acres, (4) refuse type which can generate large quantities of landfill gas (e.g., vegetation), (5) continued landfill operation (several years) for an active landfill, and (6) short time elapsed after closing for a closed landfill.[8]

If there are no customers for the electricity produced or medium/high Btu gas, energy recovery techniques are not feasible. Also, the local value of electricity and natural gas (high Btu gas) is important in choosing the energy recovery techniques. Finally, the environmental impacts of the control/treatment techniques also need to be considered. In general, internal combustion engines have the greatest secondary air impaced (e.g., NO_x, CO, and SO_x emissions) when compared to the other combustion techniques. The environmental impacts of purification techniques are a function of the specific technique used and the add-on control techniques employed.

4.2.1 Flares

4.2.1.1 Flare Process Description. Flaring is an open combustion process in which the oxygen required for combustion is provided by either ambient air or forced air. Good combustion in a flare is governed by flame temperature, residence time of components in the combustion zone, turbulent mixing of the combustion zone, and the amount of oxygen available for combustion.

4.2.1.1.1 Open flares. Flares as described in this section can be located at ground level or can be elevated. Although some of these flares

operate without external assist (to prevent smoking), most use steam or air, or the velocity of the gas itself, to mix the gas and air. Flares located at ground level can be shielded with a fence. These flares, whether or not at ground level, are described in 40 CFR 60.18. Because they cannot be easily sampled the conditions necessary to achieve 98 percent reduction are described in 40 CFR 60.18.

Landfill gas is conveyed to the flare through the collection header and transfer lines by one or more blowers. A knock-out drum is normally used to remove gas condensate. The landfill gas is usually passed through a water seal before going to the flare. This prevents possible flame flashbacks, caused when the gas flow rate to the flare is too low and the flame front pulls down into the stack.

Purge gas (N_2, CO_2, or natural gas) also helps to prevent flashback in the flare stack caused by low gas flow rate. The total volumetric flow rate to the flame must be carefully controlled to prevent low flow flashback problems and to avoid flame instability. A gas barrier or a stack seal is sometimes used just below the flare head to impede the flow of air into the flare gas network.

4.2.1.1.2 Enclosed flares. Flares described in this section are at ground level and are closely enclosed with fire resistant walls (shell) which extend above the top of the flame. Air is admitted in a controlled manner to the bottom of the shell. The temperature above the flame can be monitored and the off gas sampled. This type of flare is in use at several landfills in California and in other states. Many of these flares have been sampled and have consistently shown combustion efficiencies of greater than 98 percent for the NMOC contained in landfill gas.

The basic elements of an enclosed ground flare system are shown in Figure 4-7.[9] The landfill gas is conveyed to the flare station through the collection header and transfer lines by one or more blowers. Purge gas is usually needed only for initial purging of the system upon start-up. Landfill gas condensate is removed by a knockout drum. A water seal or flame barrier is located between the knockout drum and the flare to prevent flashbacks. The number of burner heads and their arrangement into groups for staged operation depends on the landfill gas flow rate and composition.

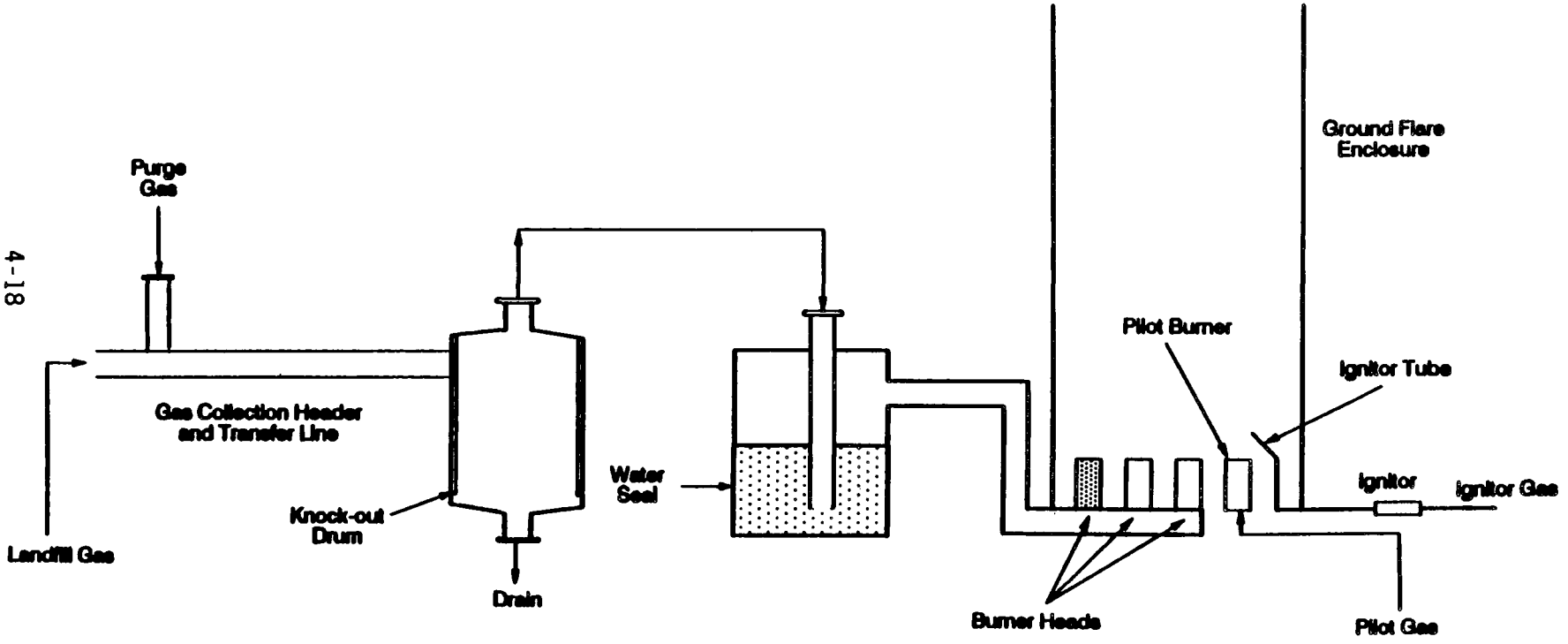


Figure 4-7. Enclosed flare.

To ensure reliable ignition, pilot burners with ignitors are provided. The burner heads are enclosed in a shell that is internally insulated. The shell can be of several shapes, such as cylindrical, hexagonal, or rectangular. The height of the flare must be adequate for creating enough draft to supply sufficient air for smokeless combustion and for dispersion of the thermal plume. Some enclosed ground flares are equipped with automatic damper controls. The damper adjusts the intake of the air by opening and closing the damper near the base of the stack, depending on the combustion temperature. A thermocouple located about 3 ft below the stack outlet is typically used to monitor combustion temperature. Stable combustion and efficient operation can be obtained with landfill gases that have heat content as low as 100 to 120 Btu/scf (or 10 to 12 percent CH_4).

4.2.1.2 Flare Combustion Efficiency. Flare combustion efficiency is a function of many factors: (1) heating value of the gas, (2) density of the gas, (3) flammability limits of the gas, (4) auto-ignition temperature of the gas, and (5) mixing at the flare tip. Combustion efficiency test data for industrial elevated flares are not readily available because of the difficulty in obtaining representative samples at the stack outlet. However, results are available from testing pilot-scale flares.[10]

The EPA has established open flare combustion efficiency criteria (40 CFR 60.18) which specify that 98 percent combustion efficiency can be achieved provided that certain operating conditions are met: (1) the flare must be operated with no visible emissions and with a flame present, (2) the net heating value of the flared stream must be greater than 11.2 MJ/scm (300 Btu/scf) for steam-assisted flares, and 7.45 MJ/scm (200 Btu/scf) for a flare without assist, and (3) steam assisted and nonassisted flares must have an exit velocity less than 18.3 m/sec (60 ft/sec). Steam assisted and nonassisted flares having an exit velocity greater than 18.3 m/sec (60 ft/sec) but less than 122 m/sec (400 ft/sec) can achieve 98 percent control if the net heating value of the gas stream is greater than 37.3 MJ/scm (1,000 Btu/scf). Air-assisted flares, as well as steam-assisted and nonassisted flares with an exit velocity less than 122 m/sec

(400 ft/sec) and a net heating value less than 37.3 MJ/scm (1000 Btu/scf), can determine the allowable exit velocity by using an equation in 40 CFR 60.18.

Unlike open flares that are not easily sampled, enclosed flares can be measured to obtain reliable test data. The effect of the surrounding environment (e.g., wind velocity) is minimized because the flare is enclosed. An enclosed ground flare burns with multiple small diffusion flames from burner heads that can be stage-operated depending on the gas flow rate. The design of enclosed ground flares allows for a wide range of combustion air flow rates and temperature control.

The SCAQMD of California requires that the flares in use at municipal solid waste landfills be the enclosed ground type flares with automatic air damper control. The SCAQMD also requires that the flare have a residence time and combustion temperature of at least 0.3 second and 1400°F, respectively. The combustion temperature is measured at 3 ft below the flare stack outlet. SCAQMD source tests for flares at municipal solid waste landfills indicate that 98 percent combustion efficiency is observed at methane concentration as low as 10 to 12 percent.

Flare NMOC emission data and combustion efficiencies for several landfills are presented in Table 4-2.[11-21]

4.2.1.3 Applicability of Flares. Flares in use at landfills for air emission control include those sites using flares as the main method of control and others using flares as a back-up to an energy recovery system. As stated earlier, the SCAQMD requires that flares in use to control air emissions at municipal solid waste landfills be the type that are enclosed with an automatic air damper control. Periodic sampling of these flares is conducted to ensure that an emission reduction of 98 percent is being achieved.

4.2.2 Thermal Incineration

4.2.2.1 Thermal Incineration Process Description. Any organic chemical heated to a high enough temperature in the presence of sufficient oxygen will be oxidized to carbon dioxide and water. This is the basic operating principle of a thermal incinerator. The theoretical temperature

TABLE 4-2. ENCLOSED GROUND FLARE COMBUSTION EFFICIENCY DATA

Landfill[b]	Date of test	NMOC Concentration (ppm) Inlet	NMOC Concentration (ppm) Outlet[c]	NMOC Mass Flow Rate (lb/yr) Inlet	NMOC Mass Flow Rate (lb/yr) Outlet	Combustion Temperature °F	Combustion Efficiency (%)	Reference
Scholl Canyon	08/01/86	3,063	.048	33.4	.005	N/A[d]	>99.99	11
	10/15/87	20,618	<.016	239	<.0012	1,400-1,500	>99.99	12
Palos Verdes (Flare Station 2)	11/16/87	51,627	<.747	2,893	<.1159	1,556	>99.99	13
Palos Verdes (Flare Station 3)	11/16/87	76.56	.67	3.8	.09	1,356	98	14
Calabasas	10/09/87	20,041	<49.7	237	<3.7	N/A	>98.44	15
	07/31/86	198	.74	2.2	.005	N/A	99.79	16
Puente Hills	12/01/87	7,065	.30	130	.05	1,710	99.96	17
Puente Hills (Flare #11)	02/20/86	6,426	40.6	8.9	.92	1,468	89.6	18
	02/21/86	5,332	53.9	9.83	1.51	1,599	84.6	19
BKK	03/04/86	19,235	22.8	61.5	.31	1,400	99.4	20
	03/05/86	8,717	82.63	79.4	1.147	1,343	98.5	21
	03/06/86	9,663	48.3	26.3	.69	1,360	97.4	21

[a]Combustion Efficiency (%) = 100 (Inlet flowrate-outlet flowrate)/(inlet flowrate)

[b]Landfill information obtained from South Coast Air Quality Management Test Reports.

[c]Outlet concentrations corrected to 3 percent oxygen.

[d]Exit flare temperature was not available.

required for thermal oxidation to occur depends on the structure of the chemical involved. Some chemicals are oxidized at temperatures much lower than others.

A thermal incinerator is usually a refractory-lined chamber containing a burner at one end. As shown in Figure 4-8, discrete dual fuel burners, inlets for the offgas, and combustion air are arranged in a premixing chamber to thoroughly mix the hot products from the burners with the offgas air streams. The mixture of hot reacting gases then passes into the main combustion chamber. This section is sized to allow the mixture enough time at the elevated temperature for the oxidation reaction to reach completion (residence times of 0.3 to 1 second are common).

Where thermal incinerators are used to control vent streams from methane recovery systems, auxiliary fuel is typically required. Thermal incinerators designed with natural gas as the auxiliary fuel may also use a grid-type (distributed) gas burner as shown in Figure 4-9. The tiny gas flame jets on the grid surface ignite the vapors as they pass through the grid. The grid acts as a baffle for mixing the gases entering the chamber. This arrangement ensures burning of all vapors at lower chamber temperature and uses less fuel. This system makes possible a shorter reaction chamber yet maintains high efficiency.

Other parameters affecting incinerator performance are the offgas heating value, the water content in the stream and the amount of excess combustion air (the amount of air above the stoichiometric air needed for reaction). The offgas heating value is a measure of the heat available from the combustion of the VOC in the offgas. Combustion of offgas with a heating value less than 1.86 MJ/Nm^3 (50 Btu/scf) usually requires burning auxiliary fuel to maintain the desired combustion temperature. Auxiliary fuel requirements can be lessened or eliminated by the use of recuperative heat exchangers to preheat combustion air. Offgas with a heating value above 1.86 MJ/Nm^3 (50 Btu/scf) may support combustion but may need auxiliary fuel for flame stability.

Combustion devices are always operated with some quantity of excess air to ensure a sufficient supply of oxygen. The amount of excess air used varies with the fuel and burner type but should be kept as low as possible.

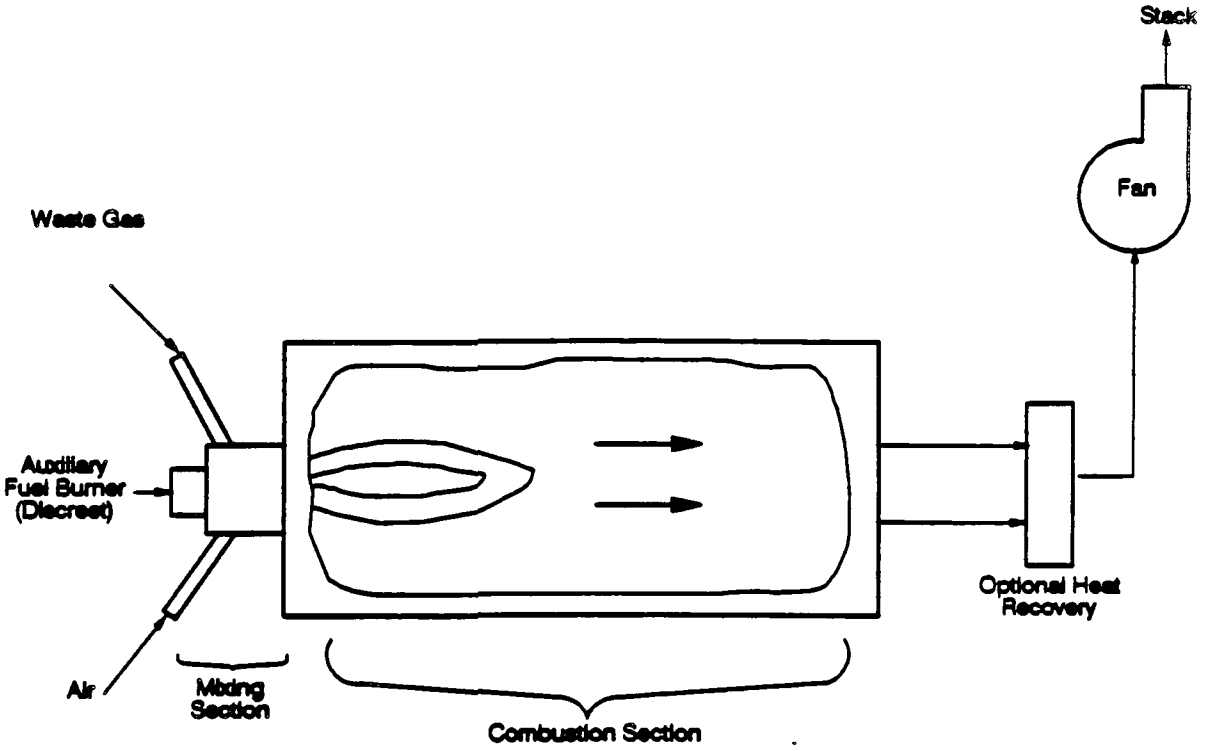


Figure 4-8. Discrete burner, thermal incinerator.

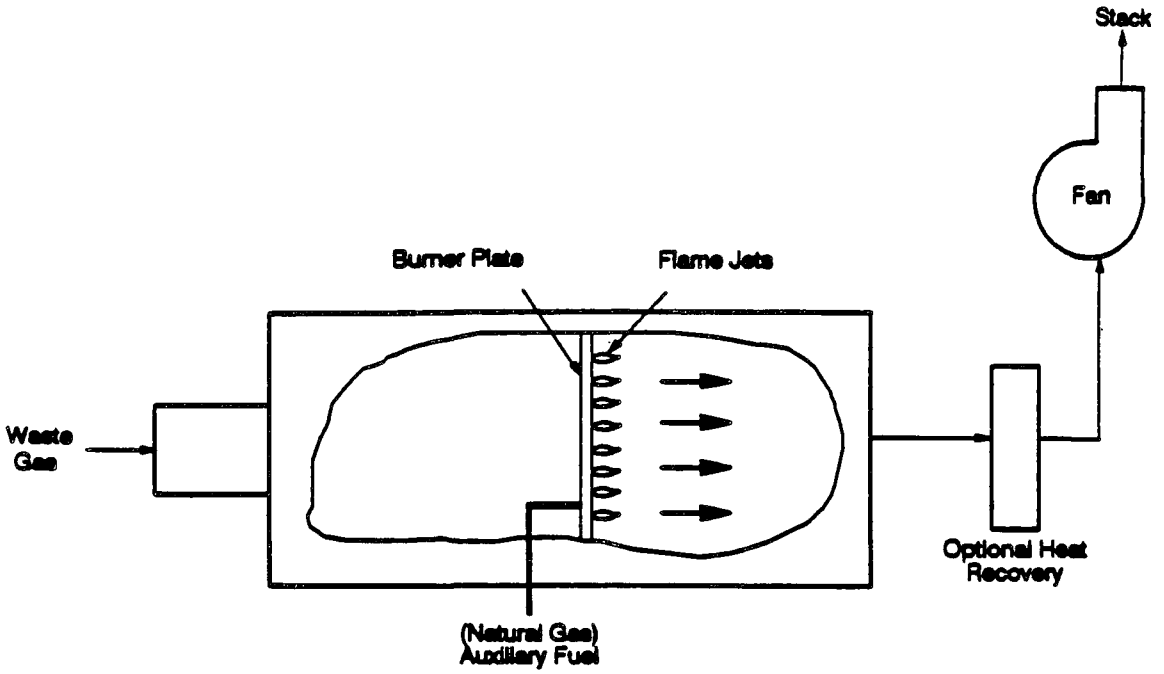


Figure 4-9. Distributed burner, thermal incinerator.

Using too much excess air wastes fuel because the additional air must be heated to the combustion chamber temperature. Large amounts of excess air also increase flue gas volume and may increase the size and cost of the system. Packaged, single unit thermal incinerators can be built to control streams with flow rates in the range of 0.1 Nm^3/sec (200 hundred scfm) to about 24 Nm^3/sec (50,000 scfm).

4.2.2.2 Thermal Incinerator Combustion Efficiency. The NMOC destruction efficiency of a thermal oxidizer can be affected by variations in chamber temperature, residence time, inlet VOC concentration, compound type, and flow regime (mixing). Test results show that thermal oxidizers can achieve 98 percent destruction efficiency for most NMOC at combustion chamber temperatures ranging from 700 to 1300°C (1300 to 2370°F) and residence times of 0.5 to 1.5 seconds.[22] These data indicate that significant variations in destruction efficiency occurred for C_1 to C_5 alkanes and olefins, aromatics (benzene, toluene and xylene), oxygenated compounds (methylethylketone and isopropanol), chlorinated organics (vinyl chloride) and nitrogen containing species (acrylonitrile and ethylamines) at chamber temperatures below 760°C (1400°F). This information used in conjunction with kinetics calculations indicates the combustion chamber parameters for at least a 98 percent VOC destruction efficiency are a combustion temperature of 870°C (1600°F) and a residence time of 0.75 seconds (based upon residence in the chamber volume at combustion temperature).[23] A thermal oxidizer designed to produce these conditions in the combustion chamber should be capable of high destruction efficiency for almost all NMOC even at low inlet concentrations.

4.2.2.3 Applicability of Thermal Incinerators. In terms of technical feasibility, thermal incinerators are applicable as a control device for any vent stream containing NMOC. In the case of landfill gas emission, however, their use is primarily limited to control of vent streams from methane recovery systems. Other NMOC destruction techniques are generally more economical for the control of landfill gas.

Incinerators can be designed to handle minor fluctuations in flows. However, excessive fluctuations in flow (upsets) might not allow the use of

incinerators and would require the use of a flare. Presence of compounds such as halogens or sulfur might require some additional equipment such as scrubbers.

4.2.3 Gas Turbines

4.2.3.1 Gas Turbine Process Description. Gas turbines take large amounts of air from the atmosphere, compress it, burn fuel to heat it, then expand it in the power turbine to develop shaft horsepower. Figure 4-10 is a simplified schematic of a gas turbine.[24] Ambient air is compressed and combined with fuel in the combustor. The combustor exhaust stream flows to the power turbine which converts some of the stream's fuel energy to rotary shaft power. This shaft power drives the inlet compressor and an electrical generator (or some other load).

Two basic types of gas turbines have been used in landfill applications: simple cycle and regenerative cycle. A simple cycle gas turbine has been described above. The gas temperatures from the power turbine range from 430 to 600°C (800 to 1,100^{o}F).[25] The regenerative cycle gas turbine is essentially a simple cycle gas turbine with an added heat exchanger. Thermal energy is recovered from the hot exhaust gases and used to preheat the compressed air. Since less fuel is required to heat the compressed air to the turbine inlet temperature, the regenerative cycle improves the overall efficiency of the gas turbine.

4.2.3.2 Gas Turbine Combustion Efficiency. The most prevalent type of gas turbine found in landfill energy recovery applications is the Solar Model Centaur. Based on a field test and information provided by the manufacturer, these turbines are capable of achieving greater than 98 percent destruction of NMOC or a 20 ppm NMOC outlet concentration at 3 percent oxygen.[26,27] Results from the only test of a Solar Model Centaur turbine showed a 6.2 ppm NMOC outlet. The NMOC destruction efficiency during this test could not be determined because the inlet NMOC concentration was not measured.

Achievement of high combustion efficiency requires the controlled mixing of fuel and air and the simultaneous satisfaction of several conditions:

- Air velocities in the combustor below flame speed.

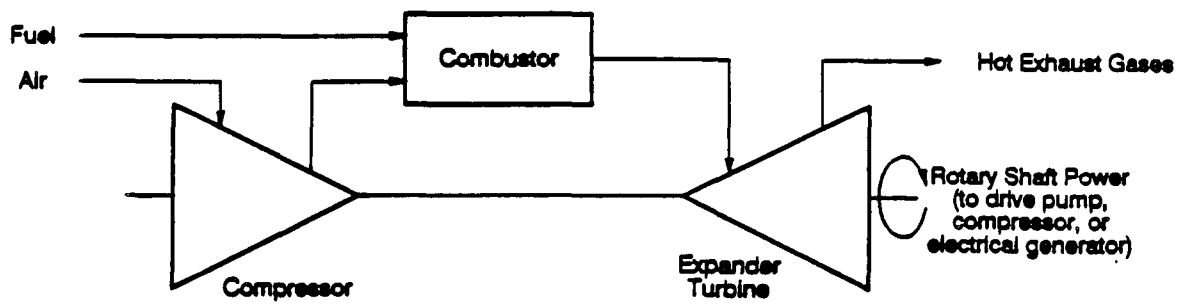


Figure 4-10. Simplified schematic of gas turbine.

- Air/fuel ratio within flammability limits.
- Sufficient residence time to complete reactions.
- Turbulent mixing of fuel/air throughout the combustion zone.
- Ignition source to start the reaction.

The heart of the gas turbine is its combustion system. Since the overall fuel/air ratio of the gas turbine is usually outside the flammable range, the combustor is divided into three zones to achieve efficient burning of the fuel. Air from the gas turbine compressor is divided and supplied to the primary combustion zone to initiate the main reaction. The reaction is mostly completed in the secondary zone. The dilution zone is used to direct the hot gases into the turbine section and reduce the temperature to meet turbine design requirements for long component life and time between inspections. Dilution is accomplished by using the correct combustor hole pattern to achieve the proper temperature profile.

4.2.3.3 Applicability of Gas Turbines. There are about 20 landfills in the U.S. which employ gas-fired turbines.[28] The applicability of a gas turbine depends on the quantity of landfill gas generated, the availability of customers, the price of electricity, and environmental issues. Gas turbines tend to have lower emissions of NO_x, CO and PM than comparably-sized internal combustion engines.

4.2.4. Internal Combustion (IC) Engines

4.2.4.1 IC Engine Process Description. Reciprocating internal combustion engines produce shaft power by confining a combustible mixture in a small volume between the head of a piston and its surrounding cylinder, causing this mixture to burn, and allowing the resulting high pressure products of combustion gas to push the piston. Power is converted from linear to rotary form by means of a crankshaft.[29]

There are two methods of igniting the fuel and air mixture: spontaneous compression ignition and spark ignition. Since spark ignition engines are typically used for in landfill energy recovery applications, only spark ignition internal combustion engines are discussed in this section. These internal combustion engines may be described by the number

of strokes per cycle (two or four) and the method of introducing air and fuel into the cylinder. In the four-stroke cycle, the sequence of events may be summarized as follows and illustrated in Figure 4-11:

- Intake Stroke--Suction of the air or air and fuel mixture into the cylinder by the downward motion of the piston.
- Compression Stroke--Compression of the air or air and fuel mixture, thereby raising its temperature.
- Ignition and Power (Expansion) Stroke--Combustion and consequent downward movement of the piston with energy transfer to the crankshaft.
- Exhaust Stroke--Expulsion of the exhaust gases from the cylinder by the upward movement of the piston.

This description applies to a naturally aspirated engine which utilizes the vacuum created by the moving piston to suck in the fresh air charge. However, many engines blow air into the cylinder with either a turbocharger or a supercharger. The turbocharger is powered by a turbine that is driven by the energy from the relatively hot exhaust gases while a supercharger is driven off the engine crankshaft. Air pressurization is used to increase the power density, or power output per unit weight (or volume) of the engine. Since the density of air increases with pressure, the mass of air that can be introduced into the cylinder increases with pressure. Furthermore, since the air-to-fuel ratio at maximum power is fixed by combustion requirements, more fuel can be introduced into the cylinder with high pressure air than with atmospheric pressure air. Therefore, more power can be obtained from a given cylinder configuration. As the air pressure is increased, its temperature is also raised. For this reason the pressurized air is often cooled before it enters the cylinder to further increase power. This process is called intercooling or aftercooling. All high power turbocharged natural gas-fueled engines are intercooled to prevent premature auto-ignition of the fuel and air mixture.

4.2.4.2 IC Engine Combustion Efficiency. The combustion or fuel efficiency of IC engines under full load is a function primarily of the air-to-fuel ratio although many other factors (such as charge homogeneity) can have an effect. As fuel efficiency decreases, emissions of nonmethane

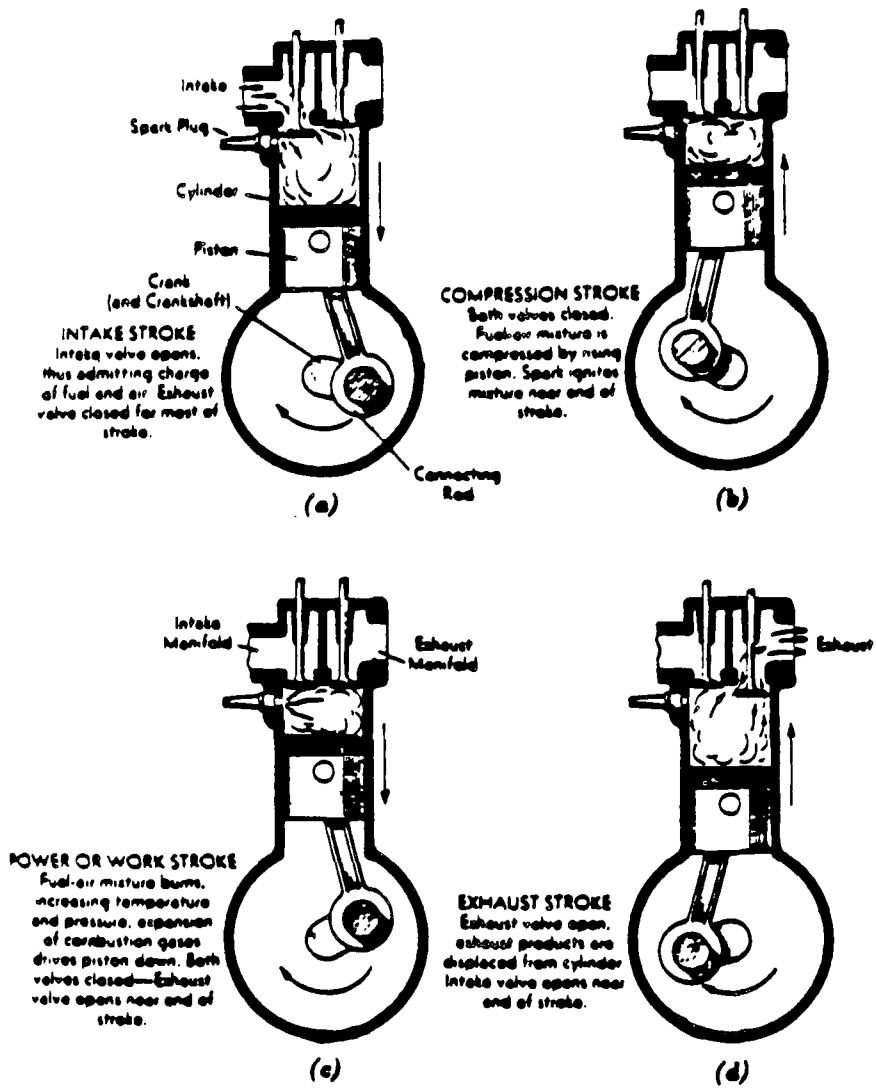


Figure 4-11. The four-stroke, spark ignition cycle. Four strokes of 180^0 if crankshaft rotation each or 720^0 of crankshaft rotation per cycle.[29]

organic compounds (NMOC) and carbon monoxide (CO), the products of incomplete combustion, increase. Minimum NMOC and CO emissions occur usually at some air-to-fuel ratio slightly leaner than stoichiometric. Below this optimum ratio, CO and NMOC emissions increase because of low temperature and insufficient oxygen for combustion. Above this ratio NMOC's increase because of low temperature.

The Environmental Protection Agency (EPA) has made a survey of the combustion efficiencies of IC engines burning various gaseous fuels including landfill gas.[30] For most of these engines only data on methane combustion efficiency is available. For these engines it is assumed that NMOC combustion efficiency will be equal to methane combustion efficiency. For a few engines NMOC combustion efficiency is known. The conclusion reached from all the information available is that IC engines can and do achieve 98 percent NMOC emission reduction at most locations. There are two situations where combustion efficiency may be less. First, if the engine is operated at reduced load, efficiencies can drop to about 95 percent. Most of these engines are operated at full load all the time. However, some engines are operated at less than full load to extend their operating life. The second factor effecting NMOC emission reduction is the fact that, in general, the State and local agencies that presently regulate internal combustion engines burning landfill gas tend to require the lowest possible NO_x, even at cost of lower engine efficiencies and higher emissions of NMOC. Some areas now require NO_x levels that result in combustion efficiencies very close to 98 percent. Internal combustion engine efficiency data for landfill gas are presented in Table 4-3.

4.2.4.3 Applicability of IC Engines. IC engines are being used at about 40 landfills because of their short construction time, ease of installation, and operating capability over a wide range of speeds and loads.[31] IC engines fueled by landfill gas are available in capacities ranging from approximately 500 KW up to well over 3,000 KW. A rule of thumb is that 1 million cubic feet of landfill gas per day at 450 Btu/scf will generate 1,250 to 1,600 KW/hr of electricity.[32]

TABLE 4-3. NONMETHANE ORGANIC AIR EMISSION DESTRUCTION EFFICIENCY - RESULTS OF FIELD TESTS OF THE COMBUSTION OF LANDFILL GAS USING INTERNAL COMBUSTION ENGINES

Landfill name/location	Information About Turbine/Generator: Type	Number and size (kilowatts)	Power output (megawatts)	Field Test Data Results: Date of test	Engine number	Outlet flow rate (dscfm)	Outlet concentration of NMHC* at 3% O_2, dry (ppmv)	References
American Canyon CA	Rich burn	two - 820	1.6	12/85	1	2,640	<2	33
				12/85	2	2,400	<2	33
Guadelupe Landfill Los Gatos, CA	Rich burn	three - 525	1.6	11/86	1	987	0.19	33
				11/86	2	744	3.8	33
				11/86	3	1,090	1.5	33
Marsh Road Menlo Park, CA	Rich burn	four - 525	2.1	09/86	1	1,420	<0.97	33
				09/86	2	1,370	<0.98	33
				09/86	3	1,460	10.1	33
				09/86	4	1,490	8.7	33
Newby Island San Jose, CA	Rich burn	four - 500	2.0	01/87	1	1,410	<1.6	33
				01/87	2	1,760	<1.5	33
				01/87	3	1,260	<1.5	33
				01/87	4	1,160	<1.5	33
Shoreline Park Mountain View, CA	Lean burn	two - 1,875	3.8	12/85	1	4,960	<1.5	33
				12/85	2	5,210	<1.6	33
City of Glendale Scholl Canyon	Rich burn	one - 1,600	0.6	01/86	2	1,424	11	34

*Concentration of nonmethane organic compounds is expressed as hexane.

4.2.5 Boilers

Boilers can be categorized into three types depending on the heat input to the furnace. Utility boilers are defined as boilers with heat input greater than 100 x 10^6 Btu/hr; industrial boilers are the boilers with heat input of 10 -100 x 10^6 Btu/hr; and domestic/commercial boilers are the boilers with less than 10 x 10^6 Btu/hr of heat input. The majority of the landfill gas-fired boilers are industrial boilers with corresponding heat inputs of approximately 10.5 x 10^6 Btu/hr (350 scfm at 50 percent CH_4) to 90 x 10^6 Btu/hr (3000 scfm at 50 percent CH_4). Therefore, the discussion of the boilers is focused on industrial boilers.

4.2.5.1 Boiler Process Description. The majority of industrial boilers are of water tube design. In a watertube boiler, hot combustion gases contact the outside of heat transfer tubes which contain hot water and steam. These tubes are interconnected by a set of drums that collect and store the heated water and steam. The water tubes are of relatively small diameter, 5 cm (2.0 inches), providing rapid heat transfer, rapid response to steam demands, and relatively high thermal efficiency.[35] Energy transfer can be above 85 percent efficient. Additional energy can be recovered from the flue gas by preheating combustion air in an air preheater or by preheating incoming boiler feedwater in an economizer unit.

When firing natural gas, forced or natural draft burners are used to thoroughly mix the incoming fuel and combustion air. In general, burner design depends on the characteristics of the fuel stream. A particular burner design, commonly known as a high intensity or vortex burner, is normally selected for gas streams with low heating values (i.e., streams where a conventional burner may not be applicable). These burners effectively combust low heating value streams by passing the combustion air through a series of spin vanes to generate a strong vortex.

4.2.5.2 Combustion Efficiency. Furnace residence time and temperature profiles vary for industrial boilers depending on the furnace and burner configuration, fuel type, heat input, and excess air level. A mathematical model has been developed that estimates the furnace residence time and temperature profiles for a variety of industrial boilers. The model predicts mean furnace residence times between 0.25 to 0.83 seconds for

natural gas-fired watertube boilers in the size range from 4.4 to 44 MW (15 to 150 x 10^6 Btu/hr).[36] Boilers at or above the 44 MW size have residence times and temperatures that ensure a 98 percent NMOC destruction efficiency. Furnace exit templates for this range of boiler sizes are at or above 12,000°C (2,200°F) with peak furnace temperature occurring in excess of 1,540°C (2,810°F). Although test data for landfill gas are not available, boilers are considered high destruction efficiency devices for NMOC present in landfill gas.

4.2.5.3 Applicability of Boilers. Landfill gas-fired boilers may be utilized in two ways. The landfill gas may be routed to an on-site boilers or piped and sold to an off-site boiler to supply heat on hot water. The landfill gas may also be routed to an on-site boiler to generate steam which in turn is fed to a steam turbine to generate electricity. The majority of landfill gas-fired boilers are utilized as a simple heat or hot water source. There is only one operating landfill gas-fired boiler to steam turbine facility in the U.S.[37] Another facility is under construction. The landfill gas-fired boiler/steam turbine system produces very little by-product emissions. However, it requires high initial capital investment and a minimum gas flow rate of 6,000 to 8,000 scfm.

4.2.6 Adsorption

4.2.6.1 Adsorption Process Description. Adsorption is a mass-transfer operation involving interaction between gaseous and solid phase components. The gas (adsorbate) is captured on the solid phase (adsorbent) surface by physical or chemical adsorption mechanisms. Physical adsorption is a mechanism that takes place when intermolecular (van der Waals) forces attract and hold the gas molecules to the solid surface. Chemisorption occurs when a chemical bond forms between the gas and solid phase molecules. A physically adsorbed molecule can be readily removed form the adsorbent (under suitable temperature and pressure conditions) while the removal of a chemisorbed component is much more difficult.[38]

The most commonly encountered industrial adsorption systems use activated carbon as the adsorbent. Activated carbon is effective in capturing certain organic vapors by the physical adsorption mechanism. In addition, adsorbate may be desorbed for recovery by regeneration of the

adsorption bed with steam. Oxygenated adsorbents such as silica gels, diatomaceous earth, alumina, molecular sieves or synthetic zeolites exhibit a greater selectivity than activated carbon for capturing some compounds. However, these adsorbents have a strong preferential affinity for water vapor over organic gases and are of little use for high moisture gas streams such as those from landfills. The landfill gas adsorption process for high Btu gas recovery consists of two major steps: (1) pretreatment removal of nonmethane hydrocarbons and water, and (2) removal of CO_2.

4.2.6.1.1 Removal of nonmethane hydrocarbons. The removal of nonmethane hydrocarbon contaminants generally requires the use of activated carbon beds. The carbon can either be replaced or thermally regenerated. Thermal regeneration of the carbon bed requires the heating of the bed with a gas stream as high as $600^{o}F$. This regeneration vent stream containing nonmethane hydrocarbons is usually incinerated in a thermal combustion chamber. An example of a pretreatment carbon bed system is shown in Figure 4-12 and the detailed pretreatment process description is given below.[39]

The landfill gas enters the adsorbent bed, and as the gas passes through the bed, the remaining water and chemical impurities are adsorbed. The resulting pre-treated mixture of methane and carbon dioxide exits the bed and is sent to the main adsorption process for further processing.

After the bed becomes saturated, and before breakthrough of any contaminants, the adsorption step is halted and feed is switched to a bed which has just completed regeneration. The breakthrough of the bed is then regenerated with hot gas to remove the chemical impurities from the adsorbent. The by-product carbon dioxide which is produced in the CO_2 removal step may be used as the hot gas. The regeneration vent stream exiting this vessel contains heavy hydrocarbons and other impurities removed from the landfill gas during the adsorption step. This effluent stream can be sent to a thermal combustor to destroy heavy hydrocarbons and other impurities.

Following regeneration of the adsorbent bed, the adsorbent must be cooled to ambient temperature prior to being placed back on adsorption. This is accomplished by passing a cool gas stream through the bed. The

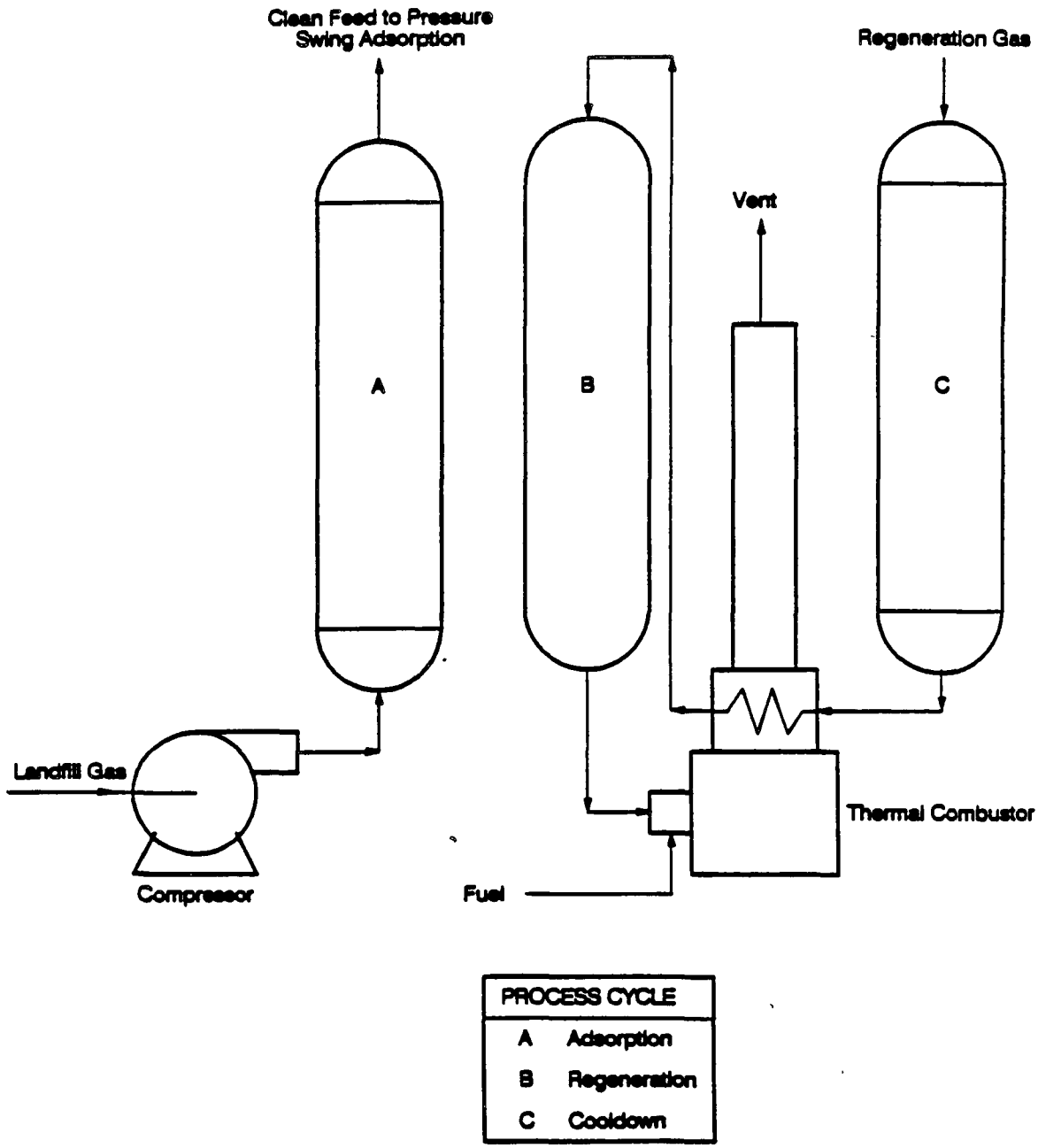


Figure 4-12. Pretreatment adsorption system.

effluent from the cool-down step is heated with the thermal combustor flue gas (if a thermal combustor is used) and then used to heat another bed. By utilizing the waste heat from the thermal combustor, the amount of fuel can be minimized.

4.2.6.1.2 Removal of CO_2. To upgrade the Btu content of the landfill gas to pipeline specifications, a minimum of ~970 Btu/scf is typically required. To meet this heat content requirement, essentially all of the CO_2 must be removed. The gas will also contain some nitrogen and oxygen which can reduce the Btu content. However, the removal of nitrogen requires extremely low temperatures that are uneconomical and impractical.[40] As a result, only the carbon dioxide is removed in upgrading the Btu content of the landfill gas. Typically, molecular sieves have been used for the removal of CO_2. The adsorption process commonly used for CO_2 removal is a pressure swing process which uses vacuum to regenerate the molecular sieve beds rather than heat. A diagram of a pressure swing adsorption process is shown in Figure 4-13 and the detailed description of a 5-step pressure swing adsorption process is given below.[41]

The pretreated landfill gas stream at feed gas pressure, combined with a methane recycle stream, enters the bottom of a bed on the adsorption step. The carbon dioxide in the feed gas is selectively adsorbed on the molecular sieve producing an exit stream of high-purity methane (99 percent) at slightly less than feed gas pressure. The adsorption step is continued until the bed becomes saturated with carbon dioxide and the mass transfer zone is just short of column breakthrough.

After the adsorption step, the values are switched and the bed is concurrently purged at feed gas pressure with a stream of high-purity carbon dioxide from the carbon dioxide surge vessel. The purpose of the high-pressure rinse step is to remove any methane which is present in the void gas or co-adsorbed on the molecular sieve following the adsorption step. The high pressure rinse step is an important feature of the process and results in greatly increased methane recovery. The purge gas which exits the bed is recycled as feed to a bed undergoing the adsorption step.

Following the high-pressure rinse step, the valves are switched and the bed which is saturated with high-purity carbon dioxide is depressurized to

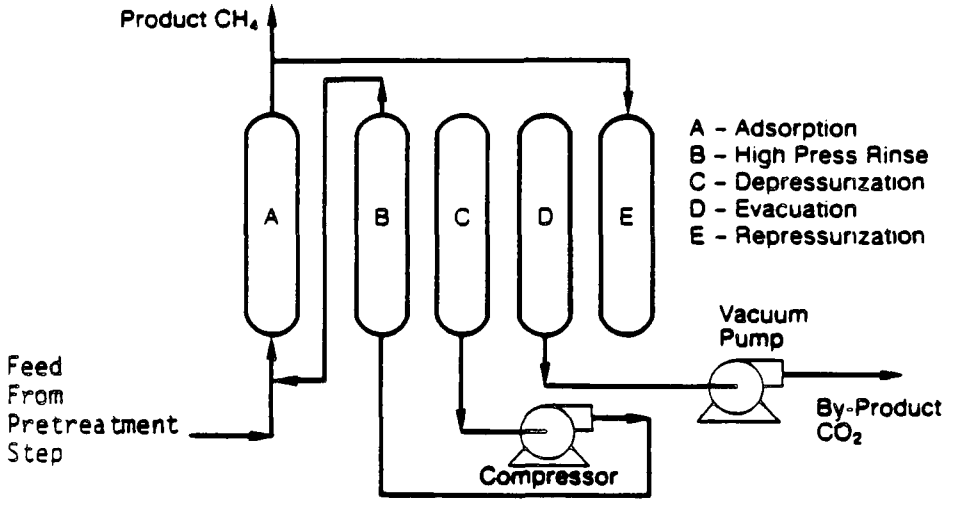


Figure 4-13. Pressure swing adsorption process.

atmospheric pressure. The desorbed carbon dioxide is recompressed to slightly above feed pressure and used as rinse for a bed undergoing the high pressure rinse step.

After the bed reaches atmospheric pressure, the valves are switched and the bed is connected to the suction of the vacuum system which reduces the bed pressure to a subatmospheric pressure. The desorbed carbon dioxide and any remaining methane are discharged at a slight positive pressure.

Following the evacuation step, the bed is repressurized to feed gas pressure with a portion of the high-purity methane produced. Repressurization is done countercurrent to the adsorption step to drive any residual carbon dioxide from the exit end of the bed. Once the feed pressure is reached, the bed is ready to repeat the cycle.

Following separation in the pressure swing adsorption process, the product methane stream may require additional compression depending on the pipeline pressure requirements. If the pipeline pressure exceeds the 80-150 psig operating range of the pressure swing adsorption process, additional product compression will be necessary.

4.2.6.2 Adsorption Control Efficiency. Control of NMOC emissions, when using methane recovery systems, is typically accomplished by routing all vent streams to a thermal incinerator. As discussed in Section 4.2.2.2, thermal incinerators are capable of achieving greater than 98 percent destruction efficiency. Therefore, routing all vent streams from the methane recovery system to an efficient thermal incinerator provides greater than 98 percent reduction of NMOC emissions.

4.2.6.3 Applicability of Adsorption Process. The feasibility of using adsorption versus other control/recovery techniques is determined by the landfill gas composition, flow rate, natural gas price, and the distance to the local gas company pipeline. Currently there are very few (two or three in the U.S.) landfill facilities which employ adsorption to recover landfill gas due to high capital investment required and low natural gas prices.

4.2.7 Absorption

4.2.7.1 Absorption Process Description. The mechanism of absorption consists of the selective transfer of one or more components of a gas mixture into a solvent liquid. The transfer consists of solute diffusion

and dissolution into a solvent. For any given solvent, solute, and set of operating conditions, there exists an equilibrium between solute concentration in the gas mixture and solute concentration in the solvent. The driving force for mass transfer at a given point in an operating absorption tower is related to the difference between the actual concentration ratio and the equilibrium ratio.[42] Absorption may only entail the dissolution of the gas component into the solvent or may also involve chemical reaction of the solute with constituents of the solution. The absorbing liquids (solvents) used are chosen for high solute (VOC or CO_2) solubility and include liquids such as water, mineral oils, nonvolatile hydrocarbon oils, and aqueous solutions of oxidizing agents such as sodium carbonate and sodium hydroxide.[43]

Devices based on absorption principles include spray towers, venturi scrubbers, packed towers, and plate columns. The control of NMOC and toxics or removal of CO_2 by gas absorption is generally accomplished in packed towers or plate columns. Packed towers are mostly used for handling corrosive materials, for liquids with foaming or plugging tendencies, or where excessive pressure drops would result form the use of plate columns. They are less expensive than plate columns for small-scale operations where the column diameter is less than 0.6 m (2 ft). Plate columns are preferred for large-scale operations, where internal cooling is desired or where low liquid flow rates would inadequately wet the packing.[44]

A schematic of a packed tower is shown in Figure 4-14. The gas to be absorbed is introduced at the bottom of the tower and allowed to rise through the packing material. Solvent flows from the top of the column, countercurrent to the vapor, absorbing the solute from the gas phase and carrying the dissolved solute out of the tower. Cleaned gas exits at the top for release to the atmosphere or for further treatment as necessary. The saturated liquid is generally sent to a stripping unit where the absorbed VOC or CO_2 is recovered. Following the stripping operation the absorbing solution is either recycled back to the absorber or sent to a treatment facility for disposal.

The solvents that can be used for the removal of water in absorption process include ethylene glycol, diethylene glycol, and triethylene glycol.

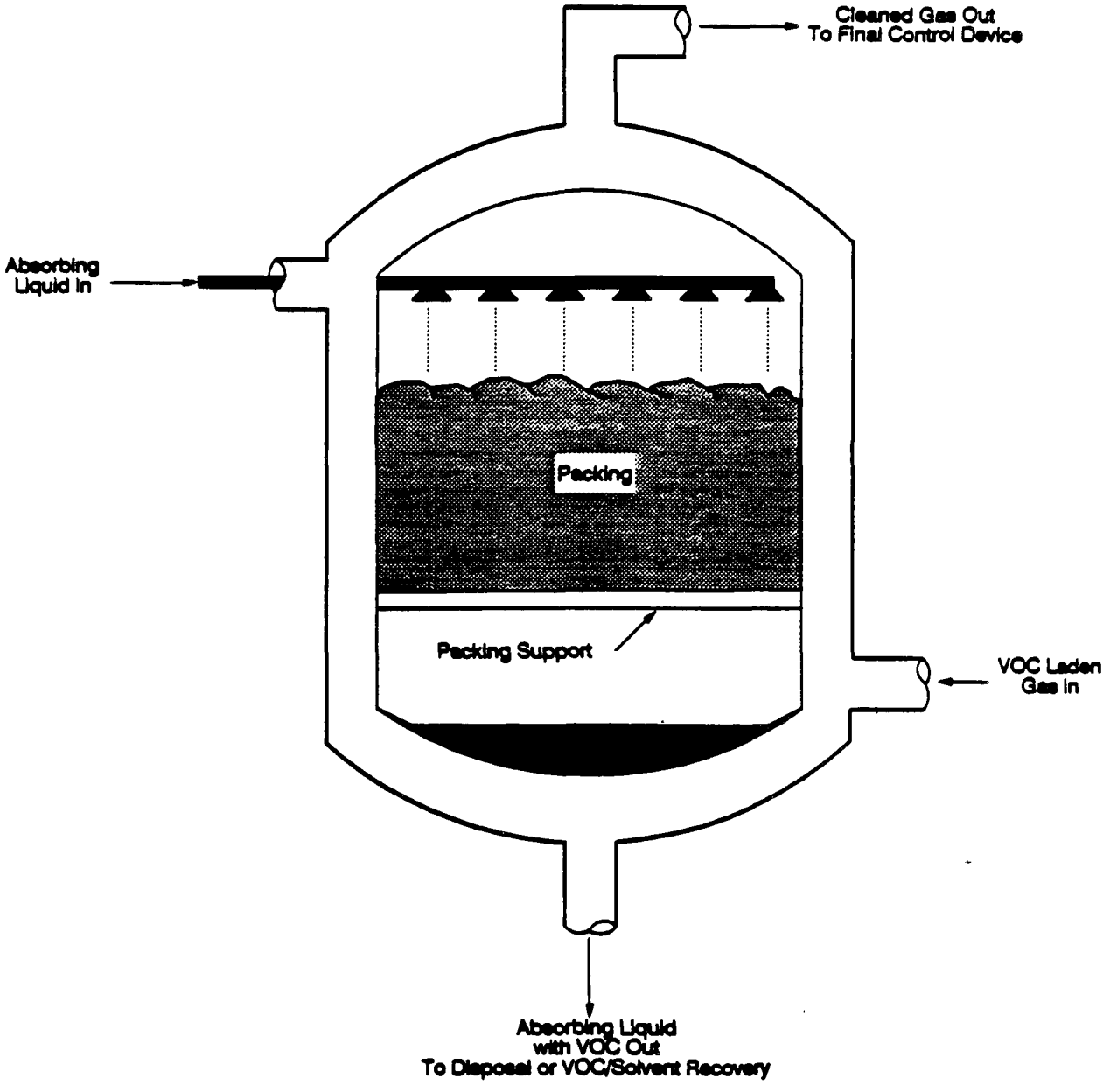


Figure 4-14. Packed tower adsorption process.

For landfill gas applications, ethylene glycol is most commonly used since diethylene glycol and triethylene glycol require high regeneration temperature. The solvents used for the removal of heavy hydrocarbons vary depending on the type of solvent selected for the removal of CO_2. Some solvents used for the removal of CO_2 also absorb heavy hydrocarbons.

The solvents used for the removal of CO_2 can be classified into the following groups; 1) organic solvents, 2) alkaline salt solutions, and 3) alkanolamines. Organic solvents include Sulfinol, Selexol, Fluor, Purisol, and Rectisol.[45] Organic solvents have an advantage over other absorption solvents because of their high acid gas loading and reduced circulation. However, organic solvents have a tendency to absorb heavy hydrocarbons thus causing faster degradation of the solvent. For a high concentration of H_2S, the Fluor and Selexol processes have been used. In the Selexol process, CO_2 is absorbed at low temperatures and high pressure. When the pressure is reduced, carbon dioxide is released. It is critical to remove as much water and heavy hydrocarbons as possible before CO_2 absorption since water and heavy hydrocarbons will reduce the affinity of Selexol for CO_2. The typical Selexol process diagram is shown in Figure 4-15.[46] The Rectisol (Methanol) process is very similar to the Selexol process except that it operates at lower temperatures.

Alkaline salt solution processes (potassium carbonate base) are applicable for treating gas with high CO_2 content, usually at pressures greater than 200 psig. Alkaline salt solution processes are not usually recommended for landfill gas treatment since most of the solvents cannot reduce CO_2 content to pipeline specifications.[47]

The alknolamine solvents include MEA (monoathanolamine), DEA (diethanolamine), and TEA (triethanolamine). An 18 percent MEA is the most commonly used solvent to remove CO_2. DEA is also used since it is noncorrosive up to 35 percent where as MEA is corrosive above 18 percent. The disadvantage of DEA is a relatively large energy requirement for regeneration.[48]

4.2.7.2 Absorption Control Efficiency. Similar to adsorption techniques, reductions in NMOC emissions are achieved by routing all vent streams to a destruction device such as a thermal incinerator. Greater than

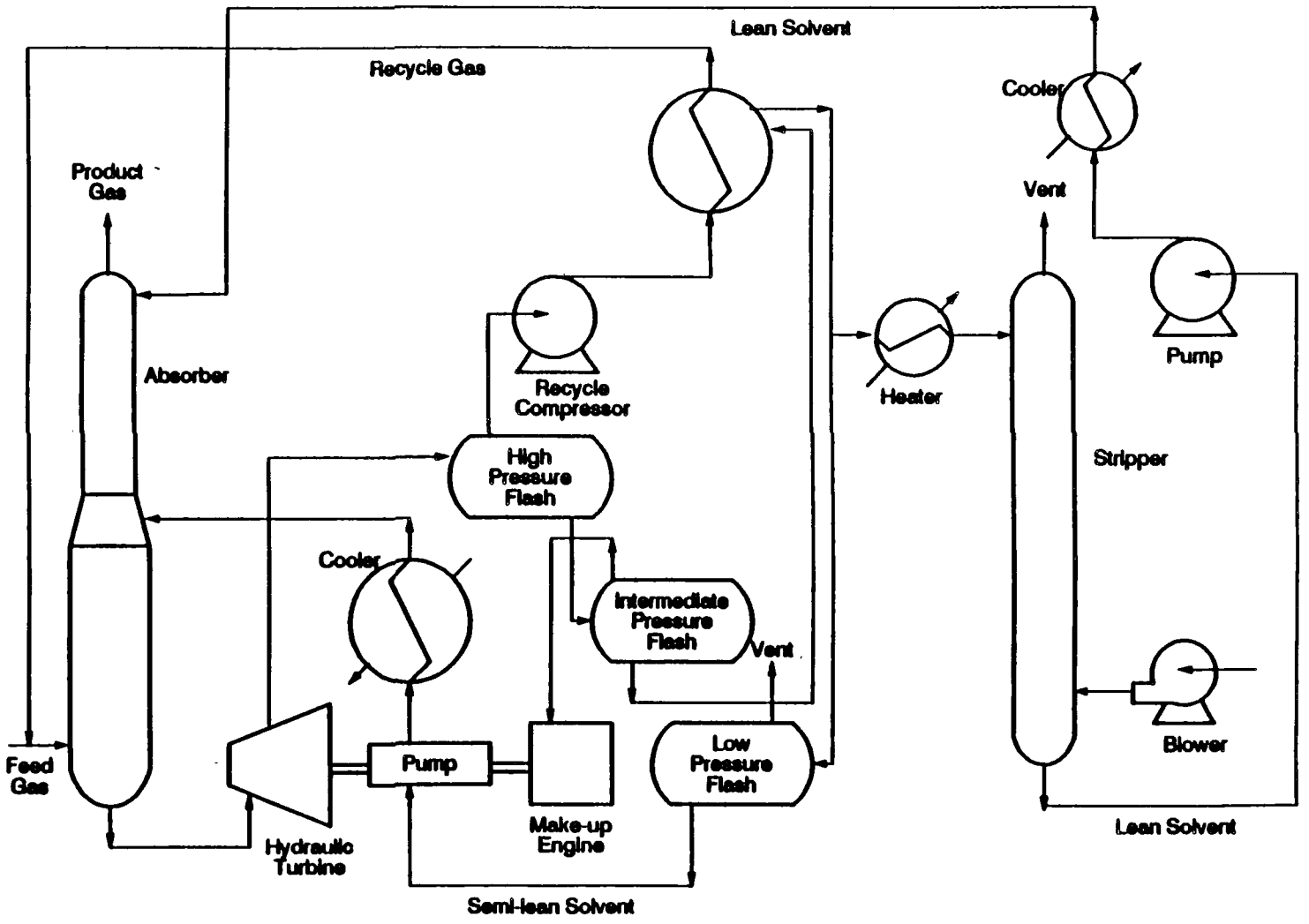


Figure 4-15. Selexol absorption process.

98 percent NMOC reduction efficiency can be achieved by routing all vent streams from the methane recovery system to a well-designed thermal incinerator.

4.2.7.3 Applicability of Absorption Process. There are few landfills in the U.S. which employ absorption (notably the Selexol process) to treat landfill gas. The applicability of an absorption process is determined by the landfill gas composition, flow rate, natural gas price, and distance to the local gas company pipeline. The absorption process also requires high initial capital investment.

4.2.8 Membranes

4.2.8.1 Membrane Process Description. Separation of gases by membrane permeation operates on the principle of selective permeability of one gas over another. The separation of carbon dioxide from a mixture of carbon dioxide and methane is accomplished by the fact that carbon dioxide permeates through the membrane much more rapidly than methane does. The result is an increase in the concentration of carbon dioxide on the low pressure side of the membrane. The methane is then concentrated on the high pressure side as the carbon dioxide is removed.

There are basically three types of membranes used commercially; 1) spiral-wound, 2) tubular, and 3) hollow fiber. The most common type of membranes used is spiral-wound, composed of cellulose acetate-based polymer. The spiral-wound membrane elements are packaged in pressure tubes. The feed gas enters the pressure tube under high pressure (500 psig), flows through the spiral-wound element and separates the gas mixture into two components; 1) low pressure permeate which contains the more permeable gas (carbon dioxide) of the mixture, and 2) high pressure residual gas which contains the less permeable component of the mixture (methane). The pressure tubes can be mounted on a skid in either a parallel or series array depending on the recovery required and flow rate of the feed gas.[49]

A typical membrane process is shown in Figure 4-16.[50] The feed gas is compressed to 500 psig, the condensed water and hydrocarbons are knocked out and/or pretreated in a carbon bed, heated to approximately 120°F, and fed to the first stage membranes which consist of three parallel pressure tubes. The feed gas is heated to 120°F for the optimal separation since membrane

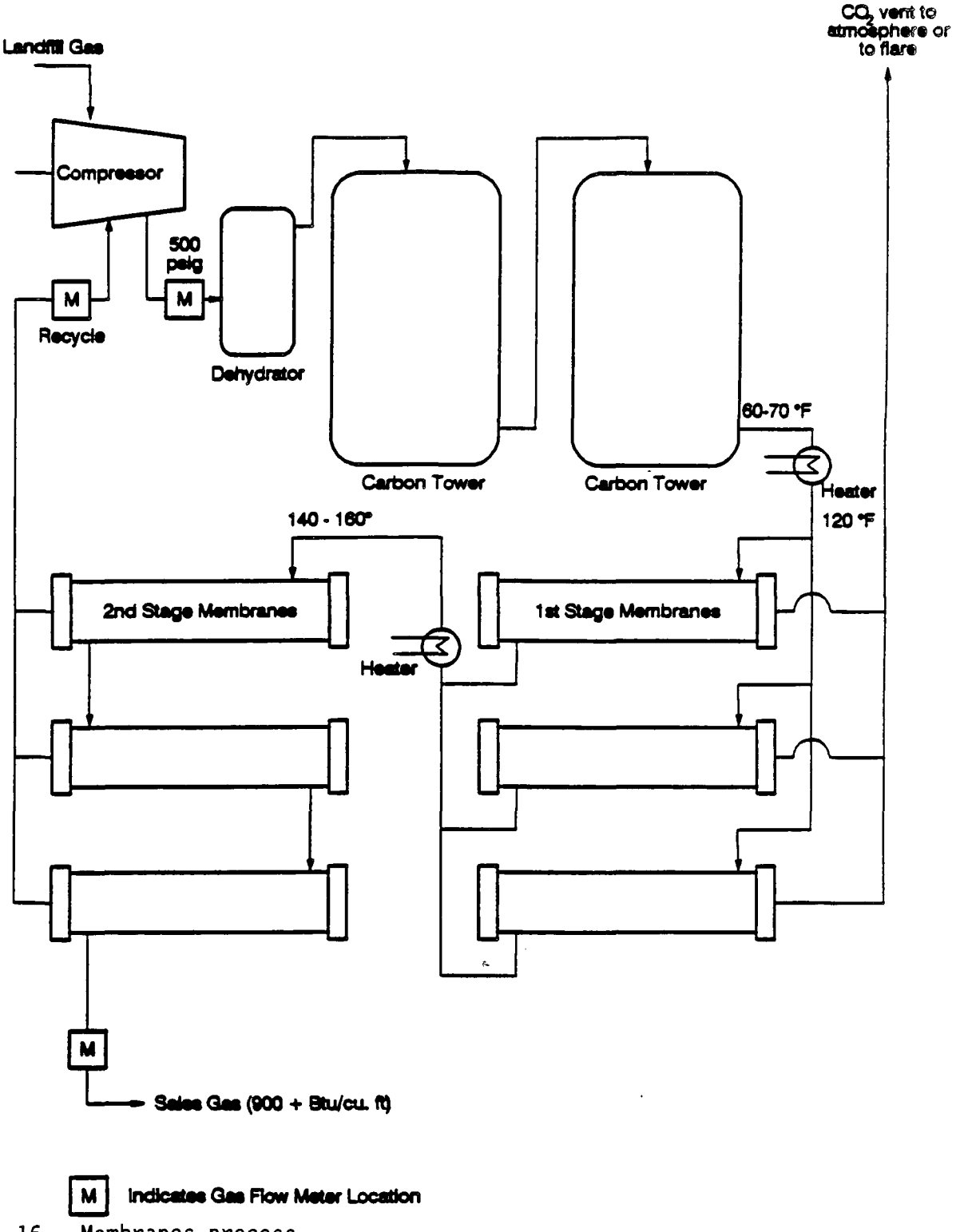


Figure 4-16. Membranes process.

pore size and gas permeability are a function of temperature. The high pressure residual gas is then reheated to 140 - 160°F and fed to the second stage membranes. The permeate gas is vented to the atmosphere or flared. The second stage membranes consist of three pressure tubes in series. The high pressure residual gas is the product stream and the permeate gas is recycled back to the feed stream. The product gas is approximately 90 percent methane. About 25 percent of the product is used as compressor fuel.

A thorough filtration is required to prevent scaling or fouling, especially for the hollow fiber membranes. These membranes are easily damaged by foreign particles and water can affect their performance. The temperature is also very critical in a membrane system. The membranes can be damaged above 160°F, and the capacity of the membranes is highly temperature sensitive.[51]

4.2.8.2 Membrane Control Efficiency. The NMOC control efficiency is dependent on the disposition of the waste gas streams (nonmethane). Depending on the heat content of the vent streams, they may be controlled by flaring or incineration. As discussed earlier, these combustion devices are capable of achieving greater than 98 percent destruction efficiency. Therefore, greater than 98 percent NMOC reduction can be achieved, if all vent streams are routed to a flare or incinerator.

4.2.8.3 Applicability of Membranes. The advantages of the membrane process are its small size, simple operation, low capital cost, and flexibility. It can handle a wide range of operating pressures, and the system can be easily modified by adding or removing the pressure tubes (in series or parallel) to adjust for the changing flow rates. However, as the methane recovery percent increases, the corresponding recovery cost also increases exponentially.

There are two landfill facilities in the U.S. which employ a membrane process.[52] The desirability of the membrane process versus other control or recovery techniques will depend on the landfill gas flow rate, the price of natural gas, the distance to the nearest gas company's pipeline, and the ratio of product gas flow rate to the pipeline flow rate. One advantage of the membrane process is its flexibility since the membrane elements can

either be added or removed to adjust for the wide range of flow rates. If the ratio of the pipeline flow rate to the product flow rate is very high, the product Btu content requirement may not be as strict due to the dilution effect.

4.3 SECONDARY AIR EMISSIONS FROM MSW LANDFILL CONTROL TECHNIQUES

This section provides a discussion of the secondary air emissions associated with MSW landfill control devices such as flares, boilers, gas turbines, and IC engines, which were discussed in Section 4.2. These control techniques themselves generate emissions in the process of controlling air emissions from MSW landfills. Consequently, EPA is concerned about the impact of these secondary emissions in evaluating the overall benefits of applying landfill air emission controls .

A summary of both the reduction and secondary air impacts associated with each of the applicable landfill air emission control devices is presented in Table 4-4. These air impacts are presented for two perspectives. The first is a very narrow perspective which considers only the air impacts at the landfill site. Emissions of particulate matter (PM), sulfur dioxide (SO_2), nitrogen oxides (NO_x), carbon monoxide (CO), carbon dioxide (CO_2), and hydrogen chloride (HCl) may be increased at the lanfill site due to operation of the control device. The second perspective is much broader and takes into account the reduction in utility power requirements and the air emission associated with electric power generation. In the case of landfill energy recovery devices such as gas turbines and IC engines, energy recovered is expected to reduce local or regional electric utility power generation. For the purpose of this analysis, electricity generated from landfill energy recovery techniques is assumed to displace an equal amount of electricity that would otherwise be generated from coal-fired utility boilers. Based on current utility fuel costs, this is a reasonable assumption. Therefore, the net secondary air impacts presented in Table 4-4 represent the difference between air emissions generated by the control equipment and air emissions that would be generated from producing an equivalent amount of electricity with a coal-fired boiler/steam turbine.

TABLE 4-4. NET AIR IMPACT FOR LANDFILL AIR EMISSION CONTROL TECHNIQUES

Control technique	Emission Reductions (lb/MM scf LFG)		Secondary Air Emissions (lb/MM scf LFG)					
	NMOC[a]	CH_4[b]	NO_x[c]	CO[d]	HCl[e]	CO_2	PM	SO_2[f]
Enclosed flare	56-3,395	21,840	4.9	58	12	60,000	Neg.	3.0
Incinerator	56-3,395	21,840	4.9	58	12	60,000	Neg.	3.0
Boiler	56-3,395	21,840	70	17	12	50,000	Neg.	3.0
(net impact)			0	0	12	0	Neg.	2.3
Gas turbine	56-3,395	21,840	26.4	12.5	12	60,000	37	3.0
(net impact)			-224	0	12	0	-15	-597
IC engine	56-3,395	21,840	111	259	12	60,000	Neg.	3.0
(net impact)			-139	0	12	0	-15	-597

[a]Estimated from NMOC concentrations found in Chapter 3 which range from 237 ppm to 14,294 ppm. Assumed a molecular weight af NMOC equal to hexane.

[b]Estimated assuming that landfill gas is 50 percent methane.

[c]Secondary NO_x air emissions for flares and incinerators are average values from the data in Tables 4-5 and 4-7, respectively. The NO_x air emissions for boilers was obtained from AP-42 for natural gas fired boilers and converted to lb/MM scf LFG assuming 500 Btu/scf. The NO_x air emissions for turbines and IC engines are average values from Table 4-8.

[d]Secondary CO air emissions for flares and incinerators are average values from the data in Tables 4-5 and 4-7, respectively. The CO air emissions for boilers was obtained from AP-42 for natural gas fired boilers and converted to lb/MM scf LFG assuming a heat value of 500 Btu/scf. The CO air emissions for turbines and IC engines are average values from Table 4-8.

[e]Secondary HCl air emissions were calculated from the NMOC compositions provided in Table 3-9 assuming all the chlorine converted to HCl.

[f]Secondary SO_2 air emissions were calculated from the NMOC compositions provided in Table 3-9 assuming all the sulfur converted to SO_2.

4.3.1 Secondary Air Emissions from MSW Landfill Control Systems

The following sections discuss the source and average amounts of secondary emissions from the control techniques discussed in Section 4.2. Factors which may impact the level of emissions of a given pollutant are also discussed. Although hydrocarbon emissions are presented, it is important to remember that the concentration of nonmethane organic compounds in the MSW landfill gas can range from 237 to 14,300 ppm, as shown in Chapter 3. The impact of secondary emissions must be considered in light of the NMOC emission reductions achieved from controlling landfill air emissions.

One factor that may impact secondary emission rates, but has not been addressed directly in calculating the emission factors presented in Table 4-4, are existing and proposed Federal and State regulations. The size of the turbines currently in use at MSW landfills is below the cutoff of the Federal regulation. However, 48 States have rules that would cover the use of gas turbines at MSW landfills. Regarding IC engines, the South Coast Air Quality Management District (SCAQMD) has regulations limiting emissions from these devices. In addition, NSPS for IC engines and small boilers have been proposed. If promulgated, these regulations would affect such devices used to control air emissions from MSW landfills. Other State and local regulations may exist. Generally, such regulations would decrease the emission levels of the criteria pollutants.

4.3.1.1 Secondary Air Emissions from Flares. As part of an EPA study, emission measurements of NO_x and hydrocarbons from a pilot-scale open pipe type (or elevated) flare were conducted.[53] The study concluded that the NO_x concentration (on an air-free basis, zero percent O_2) generally increases with increasing combustion efficiency for most flare heads and gas mixtures.

The Los Angeles County Sanitation Districts measured the NO_x and CO from enclosed flares at two of its MSW landfills.[54] As shown in Table 4-5, the NO_x emissions range from 1.4 to 10.0 lb/MM scf of landfill gas. The CO emissions range from 13.7 to 87.4 lb/MM scf of landfill gas.

4.3.1.2 Secondary Air Emissions from Thermal Incinerators. The secondary air emissions generated from thermal oxidation of landfill gas are the same ones generated from flaring landfill gas. These are NO_x, CO, and

TABLE 4-5. SECONDARY AIR EMISSIONS - RESULTS OF FIELD TESTS OF THE COMBUSTION OF LANDFILL GAS USING FLARES

Landfill name/location	Date of test	Nitrogen Oxides[a] (ppmv)	Nitrogen Oxides[a] (lbs/hr)	Nitrogen Oxides[a] (lb/MM scf LFG)	Carbon Monoxide[a] (ppmv)	Carbon Monoxide[a] (lbs/hr)	Carbon Monoxide[a] (lb/MM scf LFG)	Reference
Puente Hills	02/86	18.8	1.3	10.0	42.0	1.8	13.7	55
CA	02/86	16.0	1.4	8.1	254.0	13.3	74.9	56
BKK Corp.	03/86	7.5	2.1	2.1	172.0	5.9	28.7	57
West Covina, CA	03/86	5.0	0.3	1.4	527.0	21.2	87.4	58
	03/86	10.0	0.6	2.7	522.0	18.6	87.4	59

[a]Concentration data are expressed at 15 percent oxygen on a dry basis.

CO_2. Additionally, small quantities of PM may be generated. Also, small quantities of HCl may be generated depending on the presence of chlorinated compounds in the landfill gas. At typical thermal oxidizer combustion temperatures, essentailly all chlorine present exists in the form of hydrogen chloride (HCl).[60]

Although no data are available for thermal oxidation of landfill gas, the secondary air emissions from thermal oxidizers can be reasonably estimated from thermal oxidizer data collected from other applications. Test results from the two thermal oxidizers applied in the chemical manufacturing industry indicate that outlet NO_x concentrations, the secondary pollutant of greatest concern range from 8 to 30 ppmv.[61] This range is consistent with the NO_x emissions measured from enclosed ground flares (a very similar combustion device) burning landfill gas. Therefore, due to the lack of thermal oxidizer data and the similarity to enclosed ground flares, secondary emissions from thermal oxidizers are assumed to be the same as enclosed ground flares.

4.3.1.3 Secondary Air Emissions from Gas Turbines. The emissions generated by gas turbines burning landfill gas are those common to all combustion processes: NO_x, CO, and particulate (PM). The NO_x formation is directly related to the pressure and temperature during the combustion process. The other pollutants are primarily the result of incomplete combustion.[62]

The most important factor that affects NMOC destruction efficiency is the peak flame temperature in the primary combustion zone. Emissions of NMOC and CO increase as this peak flame temperature decreases. Also, for simple cycle gas turbines, lower pressure ratio designs tend to have higher CO and NMOC emissions than high pressure ratio designs.[63]

Nitric oxides (NO_x) produced by combustion of fuels in gas turbines are formed (mostly) by the combination of nitrogen and oxygen in the combustion air (thermal NO_x). The NO_x emissions increase with increasing peak flame temperature and increasing pressure ratios. There is, therefore, a trade off between low NO_x operation with a low peak flame temperature or a low pressure ratio and low NMOC and CO operation with high peak flame temperature or a high pressure ratio.

Small gas turbines of the size used for landfill applications are designed to meet the EPA NO_x emission limits of 150 ppmvd at 15 percent oxygen (40 CFR Part 60, Subpart GG). When landfill gas is burned in a gas turbine the resulting peak flame temperature is significantly lower than that from burning natural gas. Landfill gas turbines can be operated with NO_x levels that meet the EPA standard and in addition have combustion efficiencies greater than 99 percent. Although the landfill gas turbines in the EPA survey were below the EPA size cutoff, six of the seven turbines met the NO_x standard.[64]

A survey was conducted by EPA of the by-product emissions of gas turbines burning various gaseous fuels including landfill gas.[65] Test data for seven turbines burning landfill gas is presented in Table 4-6 and is summarized below:

- NO_x Emissions--The range in the concentration of NO_x was 11 to 174 ppmvd at 15 percent oxygen or 0.4 to 6.2 g/hp-hr. The average concentration was 44 ppmvd at 15 percent oxygen or 1.9 g/hp-hr.

- CO Emissions--The range in concentration of CO was 15 to 1,300 ppmvd at 15 percent oxygen or 0.2 to 26 g/hp-hr. The average concentration was 466 ppmvd at 15 percent oxygen or 10.4 g/hp-hr.

4.3.1.4 Secondary Air Emissions from IC Engines. The primary pollutants from landfill gas fueled IC engines are NO_x, NMOC, CO, and particulates. The NO_x formation is directly related to high pressures and temperatures during the combustion process. The other pollutants are primarily the result of incomplete combustion.

For IC engines burning most fuels, NO_x, CO, and NMOC emissions can be reduced by the use of a catalytic converter. For IC engines burning landfill gas, however, this is not possible. Various compounds from the landfill poison the catalyst resulting in loss of conversion efficiency in a few days. To change emissions for these engines it is therefore necessary to adjust the air-to-fuel ratio. Unfortunately, there is a trade-off between NO_x and NMOC emissions. Engine adjustments intended to lower NMOC

TABLE 4-6. SECONDARY AIR EMISSIONS - RESULTS OF FIELD TESTS OF THE COMBUSTION OF LANDFILL GAS USING TURBINES*

Landfill name/location	Date of test	Unit tested (#)	Secondary Air Emissions						Reference
			Nitrogen Oxides			Carbon Monoxide			
			(ppmv)	(g/hp-hr)	(lb/MM scf)	(ppmv)	(g/hp-hr)	(lb/MM scf)	
Metro Landfill Franklin, WI	04/86	1	34	a	a	NM	NM	a	66
Omega Landfill Germantown, WI	04/86	1	24	a	a	NM	NM	a	66
	04/86	2	30	a	a	NM	NM	a	66
Palos Verdes Rolling Hills, CA	03/84	1	174**	6.2	29.5	255	7.2	346	67
Puente Hills Los Angeles, CA	07/84	1	23	0.5	30.0	15	0.2	12.5	68
	08/84	2	11	0.5	25.0	294	8.1	400	68
Puente Hills Los Angeles, CA	02/86	1	11	0.4	21.2	1,300	26	1,630	69

*Concentration data are expressed at 15 percent oxygen on a dry basis.
**This represents an average of 3 runs (243, 145, 133 ppmvd at 15 percent oxygen).

NM = Not Measured

a = This value was not provided and could not be calculated because insufficient information was provided.

emissions result in increased NO_x emissions and vice versa. Although the relationship between NMOC and NO_x is complex and depends on many factors, the general relationship is illustrated below:

NO_x (g/hp-hr)	% Destruction Efficiency
2.0	98.3
5.0	98.7
10.0	99.1

The technical problem involved in reducing NO_x by increasing air-to-fuel ratio is that the extra lean mixtures are difficult to ignite and engines misfire or will not start. Engine designs overcome this problem by one or more of the following techniques.

- The use of fuel injection rather than carburetors so that all cylinders get the same mix.
- The use of indirect injection where combustion begins in a fuel rich mix in a small antechamber and travels from there to the excess air region of the main chamber.
- The use of a homogenous mix with a cratered piston to provide swirl (mixing) and a short flame path, with high voltage spark plugs.
- The use of techniques of fuel injection which result in a layer of fuel rich mix around the spark plug in the main chamber while the rest of the main chamber has excess air.

The EPA has made a survey of the secondary air emissions of IC engines burning various gaseous fuels including landfill gas.[70] Test data for 15 internal combustion engines burning landfill gas is presented in Table 4-7 and summarized below:

- NO_x Emissions--The range is the concentration of NO_x was 50 to 225 ppmvd at 15 percent oxygen or 0.6 to 3.6 g/hp-hr. The average concentration was 136 ppmvd at 15 percent oxygen or 2.4 g/hp-hr.
- CO emissions--The range in concentration of CO was 30 to 550 ppmvd at 15 percent oxygen or 0.4 to 7.2 g/hp-hr. The average concentration was 220 ppmvd at 15 percent oxygen or 2.4 g/hp-hr.

4.3.1.5 Secondary Air Emissions from Boilers. Emissions from boilers include particulate matter (PM), sulfur oxides (SO_x), nitrogen oxides (NO_x),

TABLE 4-7. SECONDARY AIR EMISSIONS - RESULTS OF FIELD TESTS OF THE COMBUSTION OF LANDFILL GAS USING INTERNAL COMBUSTION ENGINES*

Landfill name/location	Date of test	Unit tested (#)	Secondary Air Emissions						Reference
			Nitrogen Oxides			Carbon Monoxide			
			(ppmv)	(g/hp-hr)	(lb/MM scf)	(ppmv)	(g/hp-hr)	(lb/MM scf)	
American Canyon	12/85	1	59	0.8	126	218	1.8	283	71
CA	12/85	2	50	0.6	94.3	268	2.0	314	71
Guadelupe Landfill	11/86	1	68	1.4	a	271	3.5	a	71
Los Gatos, CA	11/86	2	210	2.4	a	379	2.6	a	71
	11/86	3	225	3.3	a	180	1.6	a	71
Marsh Road	09/86	1	141	3.0	a	43	0.6	a	71
Menlo Park, CA	09/86	2	156	3.2	a	89	1.2	a	71
	09/86	3	192	3.0	a	223	2.8	a	71
	09/86	4	178	3.2	a	54	0.6	a	71
Newby Island	01/87	1	159	3.6	a	30	0.4	a	71
San Jose, CA	01/87	2	103	2.8	a	276	4.6	a	71
	01/87	3	192	3.2	a	550	7.2	a	71
	01/87	4	178	3.5	a	312	3.8	a	71
Shoreline Park	12/85	1	54	0.7	95.2	211	1.6	222	71
Mountain View, CA	12/85	2	73	0.9	129	192	1.5	215	71
City of Glendale** Scholl Canyon	01/86	1	442	8.5	a	216	2.4	a	71

*Concentration data are expressed at 15 percent oxygen on a dry basis.
**The outlet is ducted to a catalytic converter for reducing NO_x emissions. The local regulatory allowable emission limit for this unit is 90 ppmvd at 15 percent oxygen.
NM = Not Measured
a = This value was not provided and could not be calculated due to insufficient information.

and lesser amounts of carbon monoxides (CO), hydrocarbons (HC), and trace elements. Nitrogen oxides are the major pollutants of concern for natural gas-fired boilers. The PM emissions factors for boilers firing natural gas or MSW landfill gas are very low because natural gas or MSW landfill gas has little or no ash content and combustion is more complete than with other fuels.[72]

The SO_x emissions from boilers are predominantly in the form of SO_2 and depend directly on the sulfur content of the fuel. The sulfur oxide emissions from boilers fired with MSW landfill gas will be negligible due to its low sulfur content. Nearly all NO_x emissions from natural gas or MSW landfill gas fired boilers are thermal NO_x. An increase in flame temperature, oxygen availability, and/or residence time at high temperatures leads to an increase in NO_x production. The rate of CO emissions from boilers depends on the combustion efficiency. For example, operation at very low excess air levels (less than two or three percent) can decrease combustion efficiency and subsequently increase CO emissions significantly.[73]

The emission factors for natural gas-fired boilers were used to estimate emissions from MSW landfill gas-fired boilers since the landfill gas mainly consists of methane and CO_2. The emission factors for natural gas-fired industrial boilers are 0.14 lb $NO_x/10^6$ Btu, 0.35 lb $CO/10^5$ Btu, and 1×10^{-3} - 5×10^{-3} lbPM/10^6 Btu.[74]

Nitrogen oxide emissions can be reduced through several operating modification such as staged combustion, low excess air firing, and flue gas recirculation. Flue gas recirculation was proven to be an effective method of reducing NO_x emissions from a MSW landfill gas-fired boiler yielding a NO_x emission factor of 0.04 lb/10^6 Btu (or 18 lb/10^6 ft).[75]

4.3.1.6 <u>Secondary Emissions from Adsorption</u>. The possible sources of secondary emissions in an adsorption system are thermal combustor flue gas (or carbon bed regeneration vent if thermal combustor is not used) and secondary CO_2 stream which may contain trace amounts of nonmethane hydrocarbons and methane. Emissions from the thermal combustor will include NO_x, SO_x, CO, and PM. The emission rates of these pollutants are a function of the design and operating parameters of the thermal combustor.

4.3.1.7 Secondary Emissions from Absorption. The possible sources of secondary emissions in an absorption process are the contaminated solvent stream and the regeneration vent. The emission rates will depend on the type of the solvent selected, design/operating parameters, and the method of treating contaminated solvent.

4.3.1.8 Secondary Emissions from Membranes. Aromatics, chlorinated hydrocarbons, and alcohols permeate with the carbon dioxide while the heavy hydrocarbons remain with the high pressure methane stream.[76] If a pretreatment system is employed to remove water and other hydrocarbon contaminants, the CO_2 vent stream will mainly be composed of CO_2 and trace amounts of methane (2 ~ 18 percent CH_4 depending on the number of membrane elements and configuration). Therefore, the major sources of secondary emissions are the CO_2 vent stream and pretreatment condensate stream. If the compressor (which compresses the feed gas before it enters the membranes) is fueled by the product gas or natural gas, the compressor exhaust also is a source of secondary emissions such as NO_x, SO_x, CO, and PM.[77]

4.3.3 The Potential for Energy Recovery Control Techniques to Reduce Demand at Utilities

In evaluating the options for control of air emissions at MSW landfills, it is important to consider the overall impact of the controls. The emission controls involving energy recovery generally yield electricity or steam. The electricity or steam produced by these controls would otherwise be produced by some other means. In the case of electricity, the net electricity generated by the MSW landfill control technique reduces the need for utility power generation. This reduction in utility requirements is likely to result in the reduction of secondary emissions from coal-fired power plants.

Under the current market conditions, demand for electricity exceeds supply. Typically, the less expensive hydro-electric and nuclear powered plants are run at maximum capacity, with the additional demand being met first by natural-gas fired plants, and then by oil and coal-fired boilers. Because the coal-fired boiler is more expensive per kilowatt, any reduction

in demand associated with generation at MSW landfills will likely replace coal-fired generation (within the constraints of grid accessibility and pre-existing contractual arrangements).

EPA judged that an analysis of secondary emissions from control techniques at MSW landfills should consider the differential between emissions from an IC engine or a gas turbine and the emissions they might "displace" at a coal-fired utility plant. The emission limits under the NSPS for coal-fired utility boilers (40 CFR 60, Subparts D and Da) are 0.03 lb PM/10^6 Btu, 1.2 lb SO_2/10^6 Btu, and 0.5 lb NO_x/10^6 Btu. These emission limits for coal-fired utility boilers were used along with the secondary emissions presented in Tables 4-6 and 4-7 for IC engines and gas turbines to estimate the net impact of control techniques involving energy recovery. These net impacts and the derivation of these net impacts is presented in Table 4-8. The emission factors for the energy recovery techniques were simply compared to the emission factors for the utility boiler to estimate relative impacts.

TABLE 4-8. DERIVATION OF NET SECONDARY AIR IMPACTS FOR GAS TURBINES AND IC ENGINES

	PM	SO_2	NO_x	HCl	CO	CO_2
Coal-fired utility boiler controlled to meet the NSPS (lb/MMBtu)	0.03	1.2	0.5	Neg.	ND	120
Reduction in coal-fired[a] utility boiler emissions per unit of landfill gas burned in a turbine or IC engine (lb/MM scf LFG)	15	600	250	Neg.	ND	ND
Secondary air emissions from a gas turbine burning landfill gas (lb/MM scf LFG)	Neg.	3.0	26.4	12	12.5	60,000
Net secondary air emissions from a gas turbine burning landfill gas (lb/MM scf LFG)	-15	-597	-224	12	0[b]	0
Secondary air emissions from an IC engine burning landfill gas (lb/MM scf LFG)	Neg.	3.0	111	12	259	60,000
Net secondary air emissions from an IC engine burning landfill gas (lb/MM scf LFG)	-15	-597	-139	12	0	0

[a]Assumed that the relative fuel-to-electricity conversion efficiencies are the same for boilers, turbines, and IC engines.

[b]Assumed the CO emissions from the combustion of coal to be negligible.

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5. REGULATORY ALTERNATIVES

This chapter describes the regulatory alternatives considered for controlling air emissions from municipal solid waste landfills. Regulatory alternatives are considered for two groups of landfills: new landfills and existing landfills. New landfills will be regulated under Section 111(b) of the Clean Air Act (CAA), while existing landfills will be controlled under the guidelines of Section 111(d). The derivation of regulatory alternatives is discussed in Section 5.1. The impact of these alternatives with respect to the number of landfills affected and the achievable emission reductions are discussed in Sections 5.2 and 5.3 for existing and new landfills, respectively.

5.1 DERIVATION OF REGULATORY ALTERNATIVES

In establishing the regulatory alternatives, the approach was taken to require air emission controls only for a subset of landfills which provides the greatest emission reduction at a reasonable costs. Controlling only a portion of the landfill population would involve establishing a cutoff (based on a site-specific characteristic) below which landfills are not required to install controls. After consideration of several regulatory formats for the cutoff, the EPA chose the annual nonmethane organic compound (NMOC) mass emission rate. Three stringency levels of this format are evaluated in this chapter: 25, 100, and 250 Mg NMOC/yr. The cutoff level of 25 Mg NMOC/yr is the most stringent, while 250 Mg NMOC/yr is the least stringent.

If a landfill's NMOC mass emission rate exceeds the cutoff before closure then gas collection and control systems must be installed. The landfill must continue to be controlled until the landfill has closed, the collection and control system has been in place for at least 15 years, and the NMOC mass emission rate falls below the same cutoff value. A cutoff is to be selected that provides the greatest emission reduction at a reasonable cost. The NMOC emission reduction for each of the stringency levels considered are discussed in Sections 5.2 and 5.3 for existing and new

landfills, respectively. The control costs for each of these stringency levels is discussed in Chapter 7.

5.2 EXISTING MUNICIPAL SOLID WASTE LANDFILLS

The OSW survey of municipal solid waste landfills described in detail in Chapter 3 was used to generate the database of existing landfills.[1] The category of existing landfills includes two types of landfills: those projected to be actively collecting waste in 1992 and those projected to have reached their design capacity and closed between 1987 and 1992. The landfills actively collecting waste in 1992 includes those landfills that would hypothetically open to replace the those closing between 1987 and 1992.

The number of existing landfills affected by the three stringency levels and the corresponding emission reduction were determined from a model which estimates the NMOC mass emission rate for each landfill in the database each year until the landfill closes, determines if controls are required and determines when controls can be removed.

Since the landfills may be affected by the cutoff at different points in time and for varying lengths of time, the series of emission reductions are the net present value in 1992 at a rate of 3 percent. The number of landfills affected by each stringency level and corresponding net present value of the emission reduction were scaled to the national level and summed to provide the total nationwide impact.

As shown in Table 5-1, approximately 1,900 landfills nationwide out of the total population of 7,480 landfills (6,034 active and 1,446 closed) would have to install controls with a cutoff of 25 Mg NMOC/yr, the most stringent option. The corresponding NMOC and methane emission reduction is about 13 million Mg NMOC and 411 million Mg CH_4 (net present value in 1992). At the least stringent cutoff of 250 Mg NMOC/yr, approximately 386 landfills nationwide (5 percent) would be affected. This stringency level would yield a nationwide NMOC emission reduction of 10 million Mg and a methane reduction of 200 million Mg (net present value in 1992).

The distribution of existing landfills affected by the stringency levels with respect to design capacity is shown in Table 5-2. At the

TABLE 5-1. REGULATORY ALTERNATIVES FOR EXISTING LANDFILLS

Stringency level (Mg NMOC/yr)	Total number of landfills affected	Percentage of total landfill population (%)	NPV of NMOC reduction (million Mg)	NPV of methane reduction (million Mg)
25	1,884	25.2	12.6	411
100	853	11.4	11.2	307
250	386	5.2	9.6	200

TABLE 5-2. DISTRIBUTION OF EXISTING LANDFILLS AFFECTED BY THE REGULATORY ALTERNATIVES

Stringency level (Mg NMOC/yr)	Distributed by Design Capacity (10^6 Mg)				Total
	<1	Between 1 and 5	Between 5 and 10	>10	
25	514	837	295	238	1,884
100	134	348	176	195	853
250	22	181	48	135	386

stringency level of 25 Mg NMOC/yr, approximately 72 percent of the total number of landfills affected are small, or less than 5 million Mg in size, while 13 percent are large, or greater than 10 million Mg in size. In comparison, approximately 35 percent of the landfill affected by the 100 mg NMOC/yr level are greater than 10 million Mg in size, while 53 percent are less than 5 million Mg in size.

5.3 NEW MUNICIPAL SOLID WASTE LANDFILLS

The number of new landfills affected by the three stringency levels and the corresponding emission reduction were estimated as described in Section 5.2 for existing landfills. [Refer to Chapter 3 for further discussion on the database of landfills and the manipulation of the information.]

Table 5-3 provides the total number of landfills affected by each stringency level and the corresponding NMOC and methane emission reduction. At the most stringent level, approximately 247 landfills are affected nationwide which is 27 percent of the new landfill population projected to be built between 1992 and 1997. The corresponding net present values of NMOC and methane emission reductions are 991,000 Mg and 51 million Mg, respectively. At 250 Mg NMOC/yr, 41 landfills would have to install controls, which would result in nationwide net present values of NMOC and methane emission reductions of 630,000 Mg and 27 million Mg, respectively.

A distribution of the new landfills affected by the stringency levels is presented in Table 5-4 with respect to landfill design capacity. Approximately 179 out of the 247 landfills affected by a stringency level of 25 Mg NMOC/yr are less than 5 million Mg, while 16 percent are greater than 10 million Mg in size. Out of the 41 landfills affected by the least stringent level, 250 Mg NMOC/yr, 24 percent are less than 5 million Mg and 42 percent are greater than 10 million Mg.

TABLE 5-3. REGULATORY ALTERNATIVES FOR NEW LANDFILLS

Stringency level (Mg NMOC/yr)	Total number of landfills affected	Percentage of total landfill population (%)	NPV of NMOC reduction (million Mg)	NPV of methane reduction (million Mg)
25	247	26.7	0.99	51
100	104	11.2	0.83	41
250	41	4.4	0.63	27

TABLE 5-4. DISTRIBUTION OF NEW LANDFILLS AFFECTED BY THE REGULATORY ALTERNATIVES

Stringency level (Mg NMOC/yr)	Distributed by Design Capacity (10^6 Mg)				Total
	<1	Between 1 and 5	Between 5 and 10	>10	
25	58	121	29	39	247
100	0	46	22	36	104
250	0	10	14	17	41

5.4 REFERENCES

1. U.S. Environmental Protection Agency, Office of Solid Waste Survey of Municipal Landfills. Data base supplied by DPRA, Inc. September 1987.

6. ENVIRONMENTAL AND ENERGY IMPACTS OF POTENTIAL CONTROLS

The environmental and energy impacts of each regulatory alternative being considered for controlling emissions from landfills have been assessed relative to baseline conditions and are presented in this chapter. Baseline conditions represent the level of control and emissions in the absence of New Source Performance Standards (NSPS) or Section 111(d) guidelines.

The impacts presented in this chapter were estimated using results of the 1987 EPA MSW landfill survey and emission estimation procedures described in Chapter 3. Impacts were calculated for each landfill in the database and the aggregated results were scaled up to yield nationwide estimates. Section 6.1 presents the estimated air impacts; Sections 6.2 addresses the potential water impacts; and Section 6.3 presents the energy impacts.

Under each of the selected regulatory options, individual landfills would be required to control landfill air emissions at different points in time and for varying lengths of time. As a result of this variability, it is difficult to assess and compare the relative impacts of each regulatory alternative without normalizing the values to some consistent basis. Therefore, all impacts quantified in this chapter are presented as net present values (NPVs) with 1992 as the base year. Future emissions are discounted using a rate of three percent. For example, 1 Mg of emissions in 1993 is counted the same as 0.97 Mg of emissions in 1992 and the same as 1.03 Mg of emissions in 1994.

6.1 AIR POLLUTION IMPACTS

The implementation of any option being considered is expected to result in significant NMOC and methane emission reductions. However, emissions of other pollutants such as NO_x, and CO (due to combustion) may be increased. Estimates of both the emission reductions and emission increase for all air pollutants of concern are presented in the following sections.

6.1.1 NMOC Emission Reductions

Under each of the regulatory options, a subset of landfills would be required to control NMOC emissions by installing and operating: (1) a

landfill gas collection system and (2) a control device which provides 98 percent destruction (and/or a 20 ppmvd outlet at 3 percent oxygen for enclosed combustion devices) for NMOC. Tables 6-1 and 6-2 present estimates of the nationwide NMOC emission reductions (expressed as NPVs) at existing and new landfills, respectively, for each regulatory alternative. The approach used to calculate NMOC emissions from MSW landfills is explained in Chapter 3. The 1992 NPV NMOC emission reductions was computed using a discount rate of 3 percent. The 1992 NPV of emission was then scaled to the national level, summed for all landfills expected to require control, and multiplied by 0.98 to reflect a 98 percent reduction.

For existing landfills, the NPV of achievable emission reductions is estimated to be 9.6 million Mg of NMOC under the least stringent nonbaseline regulatory alternative (Option 3, 250 Mg/yr cutoff). In comparison, the NPV of achievable NMOC emission reductions under the most stringent regulatory alternative (Option 1, 25 Mg/yr) is estimated to be 12.6 million Mg.

For new landfills, (i.e., those estimated to open between 1988 and 1993), the NPV of NMOC emission reductions is estimated to be 630,000 Mg under Option 3, the least stringent nonbaseline regulatory alternative. Under Option 1, the most stringent alternative, the NPV of estimated NMOC emission reductions is estimated to be 990,000 Mg.

6.1.2 Methane Emission Reductions

Landfill gas is comprised of approximately 50 percent methane, 50 percent carbon dioxide, and up to 1.4 percent NMOC, by volume. The control techniques used by regulated landfills for NMOC emissions control will also reduce emissions of methane. The NPV of potential reductions in methane are included in Tables 6-1 and 6-2 for existing and new landfills, respectively.

As shown in Table 6-1, the NPV of methane reductions are estimated to be 200 million Mg for existing landfills and 27 million Mg for new landfills under Option 3, the least stringent option. In comparison, the NPV of methane reductions are estimated to be 411 million Mg for existing landfills and 51 million Mg for new landfills, under regulatory Option 1, the most stringent option.

TABLE 6-1. NET PRESENT VALUE OF AIR IMPACTS OF REGULATORY ALTERNATIVES FOR EXISTING LANDFILLS[a]

Regulatory alternative	Stringency[b] level (Mg NMOC/yr)	Emission Reductions NMOC (10^6 Mg)	Emission Reductions Methane (10^6 Mg)	Secondary Air Emissions (10^6 Mg)[c] PM	SO_2	NO_x	CO	CO_2	HCL
1	25	12.6	411	-.29 to 0	-11.6 to .06	-4.4 to 1.4	0 to 1.1	0 to 1,200	.23
2	100	11.2	307	-.21 to 0	-8.2 to .04	-3.1 to .97	0 to .80	0 to 830	.17
3	250	9.6	200	-0.13 to 0	-5.4 to .03	-2.0 to .63	0 to .52	0 to 540	.11

[a]Air impacts are discounted at 3 percent and represented in terms of the net present value of the impacts in 1992.

[b]Stringency level reflects level above which control must be installed and below which controls may be discontinued.

[c]Ranges of secondary air emissions represent the lower and upper factors from Table 4-4. For example, the factors for NO_x range from -224 lb/10^6 scf LFG to 70 lb/10^6 scf LFG.

TABLE 6-2. NET PRESENT VALUE OF AIR IMPACTS OF REGULATORY ALTERNATIVES FOR NEW LANDFILLS[a]

Regulatory alternative	Stringency[b] level (Mg NMOC/yr)	Emission Reductions NMOC (10^6 Mg)	Emission Reductions Methane (10^6 Mg)	Secondary Air Emissions (10^6 Mg)[c] PM	SO_2	NO_x	CO	CO_2	HCL
1	25	.99	51	-.03 to 0	-1.4 to .007	-.51 to .16	0 to .13	0 to 140	.03
2	100	.83	41	-0.3 to 0	-1.1 to .006	-.41 to .13	0 to .11	0 to 110	.02
3	250	.63	27	-.02 to 0	-.72 to .004	-.27 to .08	0 to .07	0 to 73	.02

[a]Air impacts are discounted at 3 percent and represented in terms of the net present value of the impacts in 1992.

[b]Stringency level reflects level above which control must be installed and below which controls may be discontinued.

[c]Ranges of secondary air emissions represent the lower and upper factors from Table 4-4. For example, the factors for NO_x range from -224 lb/10^6 scf LFG to 70 lb/10^6 scf LFG.

6.1.3 Secondary Air Emissions

The control devices used to reduce landfill air emissions are expected to generate secondary air emissions of nitrogen oxides (NO_x), sulfur dioxide (SO_2), carbon monoxide (CO), Table 6-1. Table 6-2. particulate matter (PM), and carbon dioxide (CO_2). The estimated range of secondary air emissions for new and existing landfills is included in Tables 6-1 and 6-2 for existing and new landfills, respectively. Since the mix of control devices that would be installed under each of the regulatory options could not be accurately predicted, the secondary air emissions are presented as ranges rather than as single values. The upper end of the range represents installation of the control device with highest net secondary air emissions of that pollutant. The lower end of the range represents the net secondary air emissions of a pollutant, if all landfill owners installed the control device with the lowest secondary air emissions of that pollutant. Consistent with emission reductions, these impacts are presented as NPVs.

As shown in Tables 6-1 and 6-2, control of landfill gas emissions could actually result in decreased emissions of NO_x, PM, and SO_2. These potential reductions are based on the assumption that electricity produced from energy recovery devices will equally offset the demand for electricity at utility coal-fired generating plants. Since the emissions from combusting landfill gas are less than combustion of coal at utility generating plants per unit of energy, landfill energy recovery systems could actually reduce emissions of NO_x, PM, and SO_2.

The secondary impacts were estimated using the net emission factors from Table 4-4. A detailed discussion of these factors is provided in Chapter 4. Since landfill gas consists of approximately 50 percent methane, the NPV of methane emission reduction, in Mg, was simply converted to a volumetric gas rate using the Ideal Gas Law and then doubled to determine the NPV of landfill gas controlled. This landfill gas volume was then multiplied by the factors presented in Table 4-4 to estimate the NPV of secondary air emissions.

6.2 WATER POLLUTION IMPACTS

The main water pollution impact associated with regulating municipal landfills is the condensate formed in gas collection systems. Limited data are available on condensate formation rates. However, estimates from 3 industry sources indicate a range of about 0.01 to 0.6 gallons of condensate per scfm of landfill gas.[1-3] The condensate formed will contain a small amount of organics which may need to be treated.

6.3 ENERGY IMPACTS

Regulated landfills would be required to install a gas collection system and a gas control device. The gas collection system would require a relatively small amount of energy to run the blowers and the pumps. The gas control device would not be expected to require additional energy because the blower for the collection system is expected to maintain the air flow required by the control device. Furthermore, certain gas control devices recover energy and would contribute to a net energy savings on a nationwide basis. The NPV of energy impacts is presented in Table 6-3.

TABLE 6-3. NET PRESENT VALUE OF THE NET ENERGY IMPACTS OF EACH REGULATORY ALTERNATIVE[a]

Regulatory alternative	Stringency[b] level (Mg NMOC/yr)	Net Energy Impacts (10^6 Btu)			
		Flares		Energy Recovery[c]	
		New	Existing	New	Existing
1	25	150,000	1,200,000	7.6×10^8	6.4×10^9
2	100	120,000	880,000	6.1×10^8	4.6×10^9
3	250	77,000	570,000	4.0×10^8	3.0×10^9

[a]Impacts are presented in terms of the net present value in 1992.

[b]Stringency level reflects level above which controls are required and below which controls may be removed.

[c]Based on gas turbines at 30 percent efficiency.

6.4 REFERENCES

1. McGuinn, Y.C. Trip Report. Radian Corporation to Municipal Landfill File. January 20, 1988. Summary of November 16, 1987 visit to the Puente Hills Landfill.

2. McGuinn, Y.C. Trip Report. Radian Corporation to Municipal Landfill File. January 20, 1988. Summary of November 17, 1987 visit to the Toyon Canyon Landfill.

3. Letter and attachments from R. Echols, BFI to S. Thorneloe, EPA. October 14, 1988

7. COST OF REGULATORY ALTERNATIVES

This chapter presents the approach taken to estimate the cost of collecting and controlling air emissions from existing and new municipal landfills. There are several different control or recovery techniques that can be used to reduce air emissions from landfills. The analysis presented in this chapter evaluates an active collection system and two control techniques: one without energy recovery (i.e., flare) and one with energy recovery (i.e., gas turbine).

Section 7.1 presents the design characteristics and costs of the gas collection system. The capital and annual operating costs associated with the flare and gas turbines are presented in Section 7.2. Example costs associated with installing and operating collection and control/recovery systems can be found in Section 7.3. Section 7.4 describes the national cost impacts under the gas collection and control/recovery options.

7.1 DEVELOPMENT OF THE COLLECTION SYSTEM COSTS

This section presents the method used to develop design criteria for collecting the landfill gas. Details regarding costs for installing and operating a collection system are presented and discussed in Sections 7.1.2 and 7.1.3, respectively. Major components of a gas collection system are listed in Table 7-1 and are discussed in the following sections.

7.1.1 Collection System Sizing

Active gas collection systems consist of a multitude of extraction wells, well connectors, a gas header pipe system, a gas mover system, and a condensate collection system. Table 7-2 and 7-3 list the assumptions and equations used to conceptually design a gas collection system for cost estimating purposes.

Design of the gas collection system is based primarily on the landfill dimensions and the landfill gas generation rate. The landfills analyzed in this chapter are assumed to have equal dimensions (i.e., the length is equal to the width). This assumption is not expected to affect the cost of installing and operating collection and control equipment. The landfill gas generation rate is estimated by the Scholl Canyon Model of first order

TABLE 7-1. MAJOR COMPONENTS OF THE GAS EXTRACTION SYSTEM[a]

Item	Materials
Gas Extraction Wells	- 2 to 6" perforated piping, schedule 40 to 80, - 1" crushed stone or river gravel
Lateral Well Connections	- 10 ft PVC piping - valve - fittings
Gas Collection Header	- 3" or greater PVC piping (depending on flow/pressure requirements) - fittings
Gas Mover System	- Heavy duty, industrial type turbo blower - Variable-speed motor - valves - piping
Condensate Collecting System	- 2 to 6" PVC piping - fittings - knockout tank - pH adjustment

[a] Reference 1.

TABLE 7-2. ASSUMPTIONS USED IN DESIGNING THE GAS EXTRACTION SYSTEM

A. Gas Production

Methane generation rate: Estimated by the Scholl Canyon model[a]

Landfill gas generation rate: Twice the methane generation rate

B. Landfill Characteristics

In-place refuse density: 650 kg/m^3[b]

Operating hours: 8760 hr/yr[c]

The landfill has equal dimensions

C. Gas Characteristics

Methane concentration of the landfill gas: 50 percent[d]

Landfill gas temperature: 550^oR (90^oF)[e]

Gas velocity through the piping: 610 m/min (2,000 ft/min)[f]

Specific gravity of the landfill gas relative to air: 1.05

D. Extraction Well Design

Extraction flowrate/well: 0.04 m^3/min-m (0.4 cfm/ft) of landfill depth[g]

Default vacuum pressure at each extraction well:
1.01×10^5 N/m^2 (.9928 atm)[h]

The depth of the extraction wells is 75 percent that of the landfill depth.[i]

(continued)

TABLE 7-2. (Continued)

E. Blower System

The maximum flowrate 1 turbo blower can accommodate:
280 m^3/min (10,000 ft^3/min)[j]

Requires 30 man-hours to install the blower and motor system[k]

Retail electrical cost: $0.0511/KW.hr.[l]

F. Condensate System

Landfill gas enters collection system at 90°F and 100 percent saturation. Cools to 495°R.[m]

[a]Reference 2.

[b]Typical municipal refuse density reported in Reference 3.

[c]The extraction and control systems are assumed to operate continuously.

[d]Typical methane concentration for landfill gas reported in References 4, 5, 6, and 7.

[e]Average landfill gas temperature reported in Reference 8.

[f]Reference 9 reports that 2000 ft/min (610 m/min) is a typical gas velocity in ductwork for exhausts containing volatile organic compounds and other gaseous pollutants.

[g]Reference 10. Average extraction per well provided in References 11,12,13,and 14 divided by the average landfill depth.

[h]Typical pressure drop of extraction wells for sites visited. References 15, 16, and 17.

[i]Reference 18. Typical extraction well depth based on References 11, 12, 13, and 14.

[j]The maximum landfill gas flowrate for a turbo blower in Figure 7-5 is 10,000 cfm (280 m^3/min).

[k]Reference 19.

[l]Reference 20.

[m]Reference 21.

TABLE 7-3. DESIGN EQUATIONS FOR THE GAS EXTRACTION SYSTEM

A. Estimation of Landfill Gas Generation Rate[a]

$$Q_{lfg} = 2\ L_o\ R\ (e^{-kc} - e^{-kt}) \quad (1)$$

where,

Q_{lfg} = landfill gas generation rate at time t, m^3/yr

L_o = potential methane generation capacity of the refuse, m^3/Mg

R = average annual refuse acceptance rate during active life, Mg/yr

k = methane generation rate constant, 1/yr

c = time since landfill closure, yrs (c = 0 for an active landfill)

t = time since the intitial refuse placement, yr

B. Dimensions of a landfill based on refuse capacity.

$$L = W = \left(\frac{\text{Design Capacity}}{\rho_{refuse}\ L}\right)^{1/2} \quad (2)$$

where,

L = length of landfill, m

W = width of landfill, m

Design Capacity = design capacity of the landfill, kg

ρ_{refuse} = refuse density, kg/m^3

L = landfill depth, m

$$A = L^2 \quad (3)$$

where,

A = area of the landfill, m^2

L = length of the landfill, m

(continued)

TABLE 7-3. (Continued)

C. Radius of Influence, ROI[b]

$$ROI = (Q_w \text{Design Capacity}/\pi L\ \rho_{refuse} Q_{gen})^{1/2} \quad (4)$$

where,

R = radius of influence, m

Q_w = landfill gas flowrate per well, m^3/yr

Design Capacity = design capacity of the landfill, kg

π = 3.14

ρ_{refuse} = refuse density, kg/m^3

Q_{gen} = peak landfill gas generation rate, m^3/yr[b]

D. Landfill Pressure, P_L[c]

$$P_L = \left[\left(\frac{P_v\ (ROI^2\ \ln\ (ROI/r)\ \mu_{lfg}\ \rho_{refuse}\ Q_{gen}}{\text{Design Capacity}\ k_{refuse}\ (WD/L)} * 3.15 \times 10^{-7}\right) + P_v^{\,2}\right]^{1/2} \quad (5)$$

where,

P_L = internal landfill pressure, $Newton/m^2$

P_v = vacuum pressure, $Newton/m^2$

ROI = radius of influence, m

r = radius of outer well (or gravel casing), m

ρ_{refuse} = refuse density, 650 kg/m^3

k_{refuse} = intrinsic refuse permeability, m^2

μ_{lfg} = landfill gas viscosity, $Newton\text{-}sec/m^2$

Design Capacity = design capacity of landfill, kg

WD = well depth (i.e., 0.75L), m

L = landfill depth, m

Q_{gen} = peak landfill gas generation rate, m^3/yr

3.15×10^{-7} = conversion factor

(continued)

TABLE 7-3. (Continued)

E. Optimal Number of Extraction Wells, $Wells_{TOT}$

$$Wells_{TOT} = (\text{Landfill surface area})/(\pi R^2) \quad (6)$$

where,

$Wells_{TOT}$ = Total number of wells required

π = 3.14

landfill surface area = $(\text{length})^2$, m^2

ROI = radius of influence

F. Total Length Feet of Straight Header Pipe, H_h

$$H_h = \frac{A}{2 * ROI} + L \quad (6)$$

where,

H_h = length of straight header pipe, m

A = area of the landfill, m^2

L = length of the landfill, m

ROI = radius of influence, m

G. Diameter of Header Piping, d, for the Row of Extraction Wells[d]

$$d = \left[\frac{Q_R * 4}{(914.4\ m/min)\pi}\right]^{1/2} \quad (8)$$

where,

d = diameter, meters

π = 3.14

Q_R = flowrate due to a row of extraction wells, m^3/min

914.4 m/min = maximum gas velocity through the piping (see Table 7-2)

(continued)

TABLE 7-3. (Continued)

H. "Equivalent Length" due to standard 90° elbows[b]

$$EQ_{elbow} = [2.78\ (d * 39.37) - 1.02] * .3048 \quad (9)$$

where,

EQ = equivalent length, m, due to elbow

d = diameter of the pipe, m

I. "Equivalent Length" due to standard tees[f]

$$EQ_{Tee} = [5.82\ (d * 39.37) - 2.73] * .3048 \quad (10)$$

where,

EQ = equivalent length, m, due to tee

d = diameter of the pipe, m

J. Pressure drop across each row of header system piping[g]

$$P_2 = [(P_v * .000145)^2 - A^2 * B]^{1/2} * 6896.43 \quad (11)$$

where,

$$A = \frac{Q_R * 2118.87}{28.0\ (d * 39.37)^{2.667}}$$

$$B = \frac{Sg\ (H * 6.214 \times 10^{-4})\ T}{289}$$

where,

P_2 = exiting pressure, N/m^2

Q_R = flowrate, m^3/min

d = diameter of piping, m

Sg = specific gravity of the landfill gas

H = length of piping, m

T = landfill gas temperature, ^oK

P_v = vacuum pressure, N/m^2

(continued)

NOTE: The length of piping, H, includes the "Equivalent Lengths" associated with 1 elbow and ($wells_{TOT}$/(L/2ROI) number of tees.

K. Pressure drop across the final leg of header pipe.

$$P_3 = [(P_2 * .000145)^2 - C^2 * D]^{1/2} * 6896.43 \quad (12)$$

where,

$$C = \frac{Q_R * 2118.87}{28.0\ (d * 39.37)^{2.667}}$$

$$D = \frac{Sg\ (H * 6.214 \times 10^{-4})\ T}{289}$$

where,

P_3 = final system pressure prior to the gas mover systems, N/m^2

Q_L = 1/2 the total extraction flowrate, m^3/min

d = diameter of piping, m

H = length of piping, m

Sg = specific gravity of the landfill gas

T = landfill gas temperature, ^oK

P_2 = exiting pressure of each row of header or pipe, N/m^2

NOTE: The length of piping, H, includes the "Equivalent Lengths" associated with 2 elbows and (L/4 * ROI) number of tees.

L. Total system pressure drop, $\triangle P_{TOT}$

$$\triangle P_{TOT} = (P_L - P_3) \quad (13)$$

$\triangle P_{TOT}$ = total system pressure drop, N/m^2

M. Motor horsepower requirement[e]

$$W_{SM} = \frac{Q_{TOT}\ (\triangle P_{TOT})}{3.1536 \times 10^7\ (.65)} \quad (14)$$

(continued)

where,

W_{SM} = watt
Q_{TOT} = flowrate per blower m^3/yr
P_{TOT} = total system pressure drop, N/m^2
.65 = motor efficiency

N. Number of Blowers required[f]

$$\# \text{ Blowers} = Q_{TOT}/(283.2\ m^3/min) \tag{15}$$

where,

Q_{TOT} = total gas production rate, m^3/min
283.2 m^3/min = maximum flowrate 1 turbo blower can accomodate (see Table 7-2)

O. Condensate Flowrate, Q_{cond}[g]

$$Q_{cond} = \frac{.0203\ Q_{TOT}}{760 - 1.87 \triangle P_{TOT}} \tag{16}$$

where,

Q_{cond} = flowrate of condensate, m^3/min
Q_{TOT} = total gas production rate, m^3/min
$\triangle P_{TOT}$ = total system pressure drop, N/m^2

[a]Reference 22, p. 8.

[b]The peak landfill gas generation rate can be estimated using equation (1) with t equal to the landfill age at closure.

[c]Reference 23, p. 202.

[d]Reference 24, p. 3-3.

[e]Reference 25.

[f]Assumed that one blower can process up to 10,000 cfm (283.2 m^3/min) of landfill gas.

[g]Reference 26.

decay.[27] Equation 1 in Table 7-3 gives the form of the model used to estimate the landfill gas generation rate. A detailed discussion of this model can be found in Chapter 3.

7.1.1.1. Gas Extraction Well. The gas extraction wells are assumed to be installed within the interior, or refuse fill, of the landfill. Vertical extraction wells (12 to 36 inch diameter) are excavated and back filled with 1 inch or larger crushed stone and 5 to 15 centimeters (2 to 6 inches) in diameter perforated PVC piping.[28] In this cost analyses, the depth of the extraction wells are assumed to be 75 percent of the landfill depth to insure that the well will not puncture the landfill lining or interfere with a leachate collection system when installed.

A design parameter referred to as the "radius of influence" is estimated to determine the number of extraction wells required to cover the entire landfill area. The radius of influence is the maximum distance that a well can extract a gas molecule by means of a pressure differential. The radius of influence, R, can be estimated using Equation 2 found in Table 7-3.

The number of extraction wells can be estimated by dividing the landfill area at capacity by the area that one extraction well can influence. This "area of influence" is simply R^2. This approach estimates the maximum number of wells required to extract all of the landfill gas that is expected to be generated. The gas extraction rate for each extraction well is assumed to be 0.04 m^3/min-ft (0.4 cfm/ft) of landfill depth.[29]

7.1.1.2 Lateral Well Connections. The number of lateral well connections is equal to the number of extraction wells. Included in the well connection is a control valve, 3 meters (10 feet) of PVC piping, and a monitoring port with cap.

7.1.1.3 Header System. Each extraction well is connected to a 15 to 70 centimeter (6 to 27 inch) diameter PVC header pipe system. The header pipe system is laid out to convey a vacuum from a gas mover system to the wells and in turn transport the landfill gas to an emission control device. The configuration of the header system realistically depends on the landfill perimeter configuration.

The total length of PVC header pipe required to conduct the vacuum and transport the landfill gas can be estimated by using Equation 5 presented in Table 7-3. The gas extraction wells in this cost analysis are placed in straight rows, spaced at a distance of two times the radius of influence. Each row of extraction wells is connected to an adjacent header pipe which converges to a final, larger diameter pipe. This final header pipe is referred to as the "final leg". All of the adjacent header pipes converge upon this final leg. The header pipe system is assumed to be installed on the surface of the landfill. Figure 7-1 illustrates the header pipe system layout used in this cost analysis.

7.1.1.4 Gas Mover System. The blowers and motors used to transport the exhaust gas to the emission control device are sized for the maximum volume of landfill gas that is expected to be produced during the functional life of the gas mover system (15 years). The pressure drop due to piping across the entire collection system and the total gas production rate of the landfill are functional parameters required to determine the size of gas moving equipment. The components of a gas moving system include a heavy-duty, industrial type, turbo blower(s) and variable-speed motor(s). It is assumed that the blower(s) can be idled down to accommodate the landfill gas production rate as it changes through the years of operation.

Equations 9 and 10 in Table 7-3 are used to estimate the pressure drop across the header pipe system (excluding the extraction wells). A number of assumptions and calculations are made in order to use Equations 9 and 10. Such calculations include the header pipe diameters and "equivalent length" estimations for standard elbows and tees. It is assumed that the landfill gas temperature is $550^{o}R$ ($90^{o}F$) with a gas molecular weight of 30. The pressure is assumed to be 1.01×10^{5} Newton/m^{2} (0.9926 atm) exiting each extraction well. The total system pressure drop is the sum of the total pressure drop across the header system and extraction wells. Refer to Figure 7-2 for a graphical interpretation of the system pressure drop.

A flow rate of 280 m^{3}/min (10,000 cfm) of landfill gas is assumed to be the maximum volumetric flow rate that a single blower can accommodate. Therefore, the number of blowers, and the number of motors, can be estimated by applying Equation 13 in Table 7-3. Once the system pressure drop and

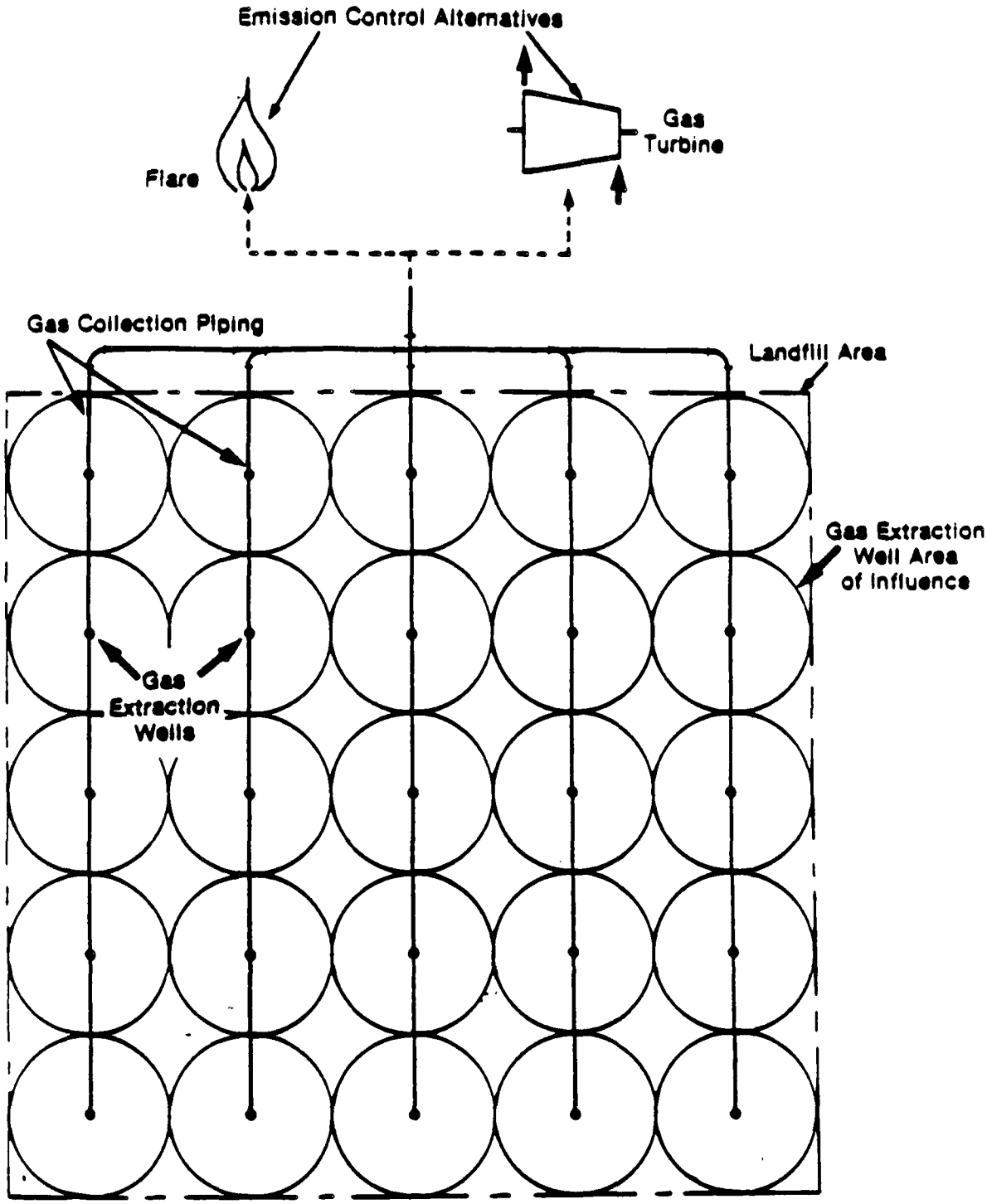

Figure 7-1. Theoretical header pipe system.

peak landfill gas flow rate during the motor(s)' functional life are determined, Equation 12 in Table 7-3, is used to determine the horsepower requirement of the motor(s).[30]

7.1.1.5 Condensate System. Condensation of the landfill gas usually occurs on the inside of the header pipe system due to the cooler temperatures at the surface of the landfill. The condensing landfill gas vapor consists mainly of water; however, it also contains trace amounts of nonmethane organic compounds. A typical condensate disposal procedure used is to collect the condensate in a knock-out tank, adjust the condensate's pH by adding caustic at the landfill facility prior to discharge to a public water treatment facility.

In this design analysis, the knock-out pot and the pH adjustment facility are sized based on the maximum expected landfill gas flow rate. The amount of condensate PVC piping, usually 5 to 10 centimeters (2 to 4 inches) in diameter, is estimated to be 4 percent of the header pipe requirement.[31] Two equations were derived to express the installed capital and the annual operating cost of the condensate system as a function of the landfill gas flowrate. These equations are based on documented equipment purchase costs and information regarding leachate disposal from landfills.[32,33] These equations are presented in Section 7.1.2.5.

7.1.2 Capital Cost Bases

The equations and bases for the capital costs of the equipment required to collect landfill gas are presented in Table 7-4. The capital cost represents the total financial resources required to plan, engineer, install, and test run the collection system. These costs are segregated into direct and indirect costs. The direct capital cost includes the investment required to purchase and install the extraction wells, well connections, header system, gas mover system, and condensate system. The indirect costs include engineering, contractor's fee, construction fee, start-up, performance test, model study, and contingencies. Typically, direct and indirect costs for fixed-capital investments are percentages of the purchased equipment cost, ranging from 15 to 40 percent. The cost factors presented in Table 7-5 are similarly applied to the purchase cost. The purchase cost is assumed to be 60 percent of the direct capital cost.

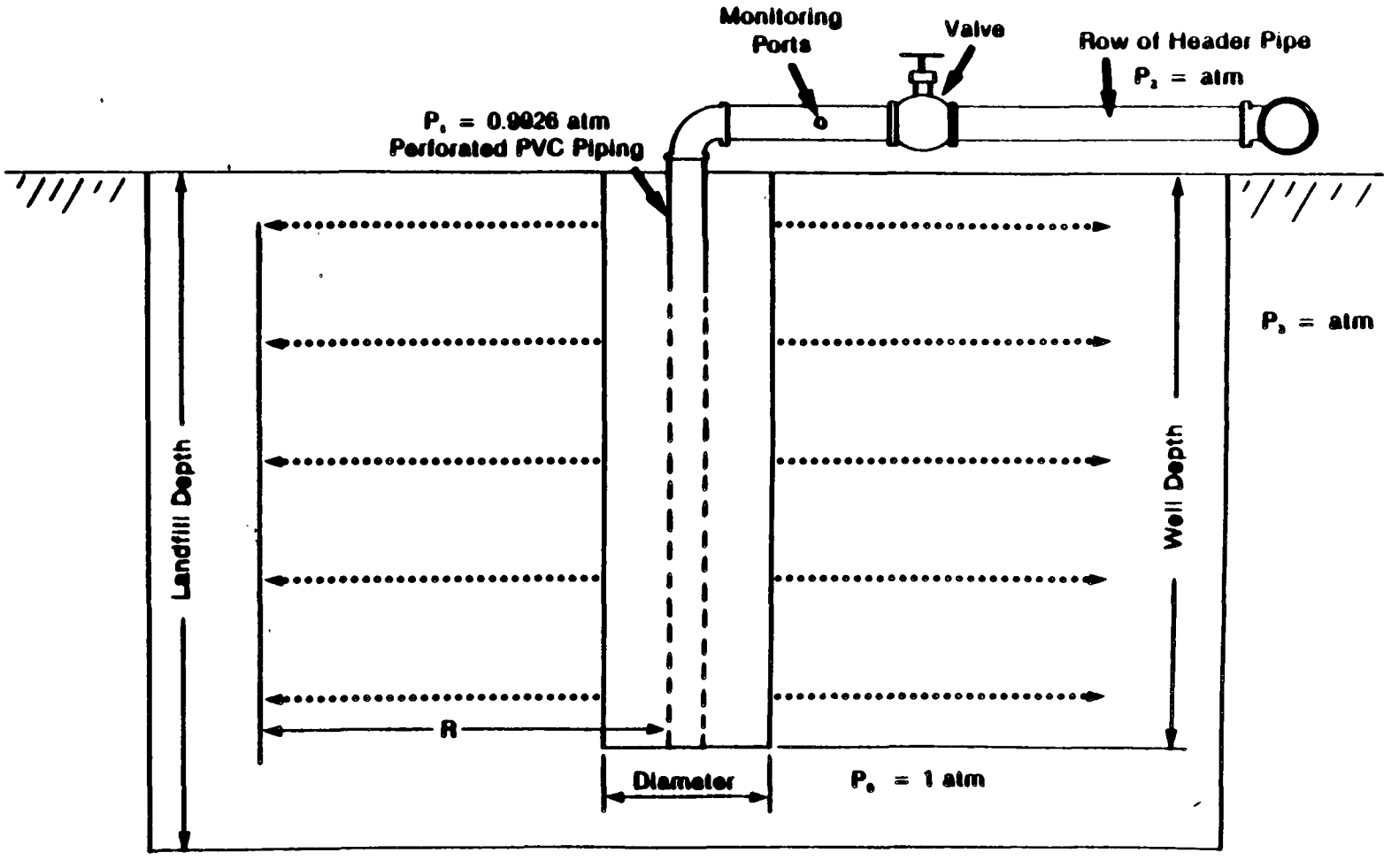

Figure 7-2. Graphical representation of the system pressure drop.

TABLE 7-4. CAPITAL COST BASES AND EQUATIONS FOR THE GAS COLLECTION SYSTEM

Item	Unit	Cost per unit/bases
A. Gas Extraction Well[a,b]	Vertical meter (greater than 24.4m)	$205.00 (1985) $410.00/meter over 24.4m
B. Lateral Well Connections[a]	Each	$1,250.00 (1985)
C. Gas Collection Header[a]	Linear meter	$65.60
D. Gas Mover System[c]	Each	
	a) Blower (see Figure 7-4)	
	b) Motor (see Figure 7-5)	
E. Condensate Collection System		
Piping[d]		.04 ($ Header)
Knockout pot[e]		$3,000
pH adjustment[f]		$\$22,500 \left(\frac{Q_{cond}}{2.79}\right)^{0.6}$

Equations Used to Estimate the Total Item Costs

Extraction Wells

$ Wells = Total number of wells (well depth, m) ($205.00/m) (1)

Lateral Well Connections

$ Connections = Total number of wells ($1,250.00/well) (2)

Gas Collection Header

$ Header = Total length (m) ($65.60/m) (3)

(continued)

TABLE 7-4. (Continued)

Item	Unit	Cost per unit/bases
Gas Mover System		
- Blower = Figure 7-4		
- Motor = Figure 7-5		
Condensate Collection System		
$ Condensate = 0.04 ($ Header) + 3,000 + 22,500 $\left(\frac{Q_{cond}}{2.79}\right)^{0.6}$		(4)

[a]Reference 34, p. 6-25. Price quoted at $62.50/ft.

[b]Reference 35.

[c]Reference 36, p. 562.

[d]Reference 37.

[e]Reference 38.

[f]Reference 39, p. 7-9.

TABLE 7-5. DIRECT AND INDIRECT CAPITAL COST FACTORS FOR THE COLLECTION SYSTEM[a]

	Extraction wells	Well connection	Header system	Gas mover blower	System motor	Condensate system
Direct Cost Factors						
Purchase	b	b	b	d	e	f
Taxes	*	*	*	.03	.03	.03
Freight	*	*	*	.05	.05	.05
Installation	c	c	c	352.00	*	*
Foundation	*	*	*	.12	.12	.12
Erection	*	*	*	.40	.40	.40
Electrical	*	*	*	.01	.01	.01
Piping	*	*	*	.02	.02	.02
Building	*	*	*	.40	.40	.40
Indirect Cost Factors						
Engineering and supervisors	.10	.10	.10	.10	.10	.10
Construction and field expenses	.10	.10	.10	.10	.10	.10
Construction fee	.10	.10	.10	.10	.10	.10
Start-up	.01	.01	.01	.01	.01	.01
Performance test	.01	.01	.01	.01	.01	.01
Model study	.02	.02	.02	.02	.02	.02
Contingencies	.03	.03	.03	.03	.03	.03

*Included in installed costs
**Included in blower installation costs

[a]Reference 43, p. 3-11; Reference 44

[b]60% of the estimated installed cost

[c]40% of the estimated installed cost

[d]Obtained from Figure 7-4

[e]Obtained from Figure 7-5

[f]Obtained from Equation 4 in Table 7-4

NOTE: All cost factors applied to the purchase price.

7.1.2.1 Gas Extraction Well. The direct capital cost of one gas extraction well in 1985 dollars is estimated to be $205.00 per vertical meter up to 24 meters ($62.50 per vertical foot up to 80 feet). For wells greater than 24 meters, the rate converts to approximately $410 per vertical meter beyond 24 meters. The direct capital cost is escalated to 1987 dollars using the piping cost indices reported in Chemical Engineering[40] and presented in Table 7-6. Typical percentages of fixed-capital investment values for direct and indirect costs range from 15 to 40 percent of the purchased equipment cost.[41] Therefore, all the indirect cost factors presented in Table 7-5 are applied to the purchase price which is assumed to be 60 percent of the direct capital cost. The total installed capital cost of gas extraction wells is simply the product of the total number of wells required, the extraction well depth, and the total installed capital cost (including direct and indirect capital).

7.1.2.2 Lateral Well Connection. The 1985 direct capital cost of a lateral well connection is estimated to be $1,250.00 each. This value is the median value reported for lateral well connections in Reference 1. The 1985 direct capital cost is escalated to 1987 dollars using the indices presented in Table 7-6. The total installed cost of lateral well connections is merely the product of the total number of extraction wells and total installed capital cost per connection.

7.1.2.3 Header System. The direct capital cost per linear foot of PVC piping, including all appropriate fittings, for landfills less than or equal to 5 million tons of refuse at capacity is estimated to be $66.00 per linear meter in 1985.[42] This figure is the lower end value reported in the Reference 2 for gas collection headers. Equation 3 in Table 7-4 is used to estimate the direct capital cost per linear foot for those landfills with a refuse capacity greater than 5 million tons. The 1985 direct capital cost is escalated to 1987 dollars using the factors in Table 7-6. The total installed capital cost of the header system is the product of the length of header required and the total installed capital cost per linear foot.

7.1.2.4 Gas Mover System. Figures 7-3 and 7-4, obtained from Reference 43, are used to estimate the 1979 purchase price for the heavy duty blower(s) and variable-speed motor(s), respectively. The purchase

TABLE 7-6. COST INDEX

Equipment item	CE index	Base year index	August 1987 index
Extraction Well	Pipe	August 1985 - 384.3	388.8
Well Connection	Pipe	August 1985 - 384.3	388.8
Header Pipe	Pipe	August 1985 - 384.3	388.8
Blower	Pumps	August 1979 - 284.5	431.0
Motor	Pumps	August 1979 - 284.5	431.0
Condensate System	Pipe	August 1985 - 384.3	388.8
Turbine System	Equipment	August 1983 - 335.9	344.9

Source: Reference 45.

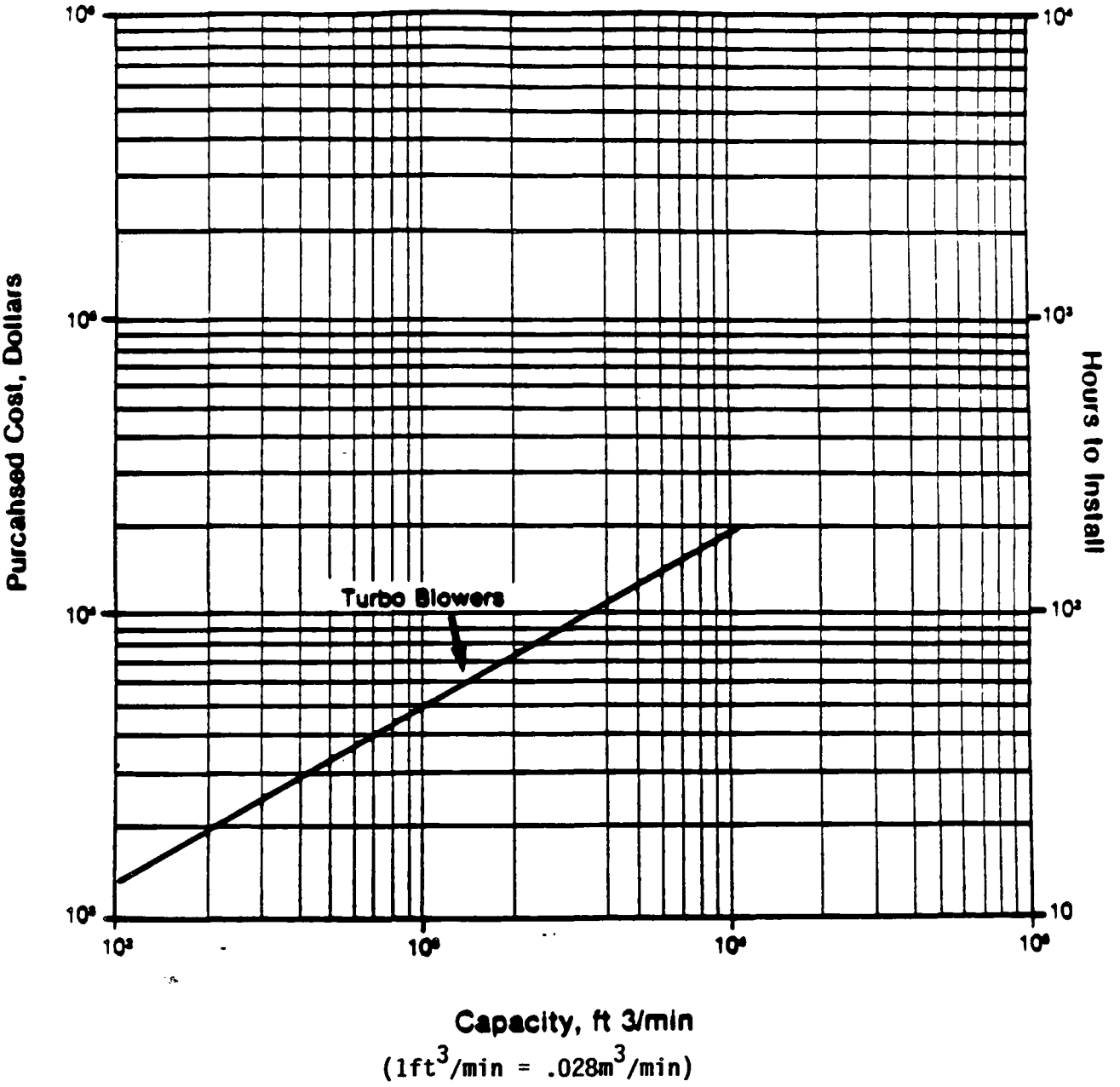

Figure 7-3. Blowers purchase price (1979 dollars).[48]

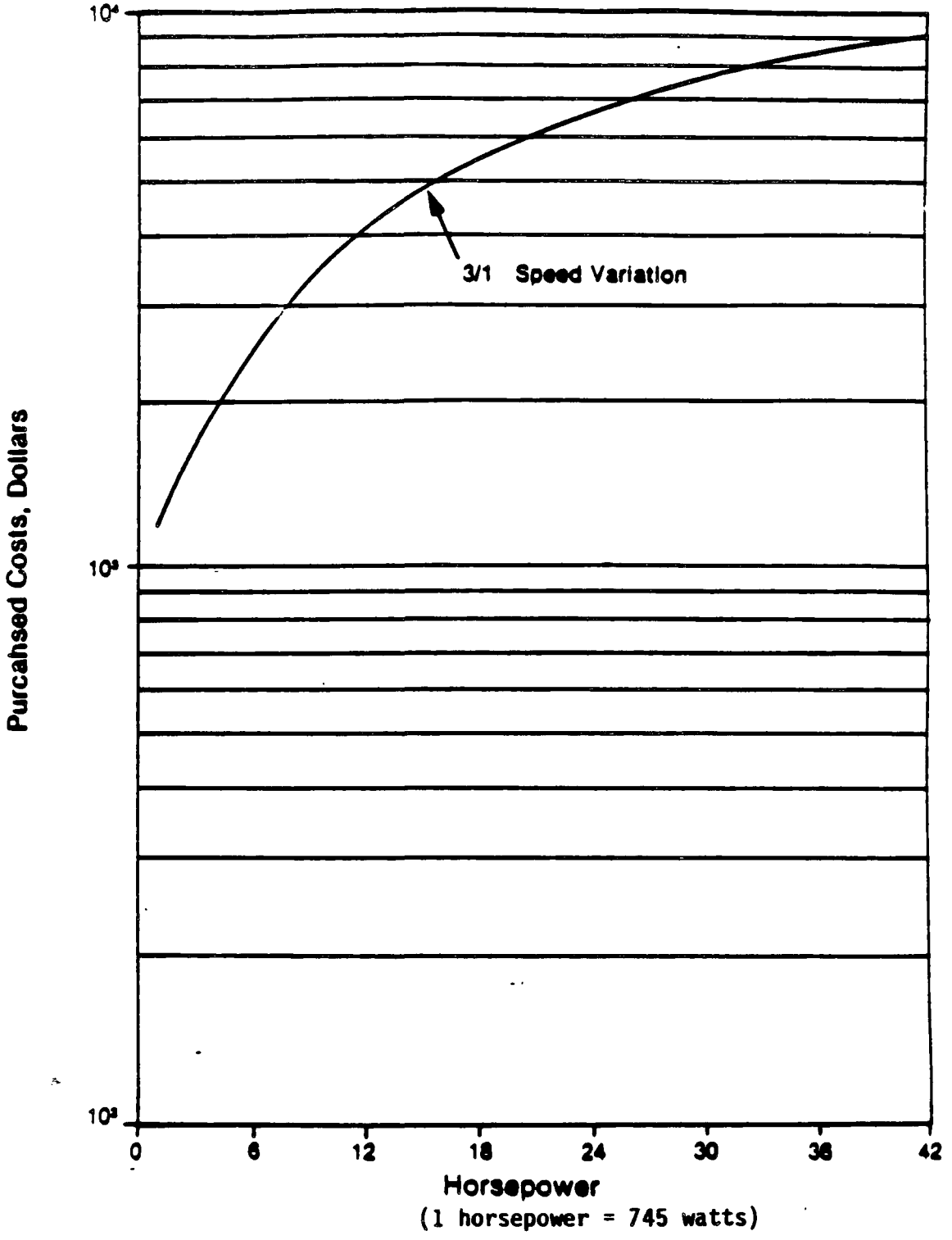

Figure 7-4. Motor purchase costs (1979 dollars).[49]

price of each is escalated to 1987 dollars using the pump cost index reported in Reference 47. The total capital cost for the initial gas mover system is estimated based on the maximum landfill gas flow rate expected during the functional life of the system. The gas mover system is assumed to require 30 man-hours of labor to install each blower and motor combination.[46] The appropriate tax and freight charges are applied to the purchase price of the blower(s) and motor(s) in addition to the remaining direct installation factors presented in Table 7-5.

7.1.2.5 Condensate System. Equation 4 in Table 7-4 is used to estimate the capital cost of the condensate system. The condensate collection system includes a knockout pot, a pH adjustment system, and piping. Costs for the individual components are also provided on Table 7-4. Factors in Table 7-5 are used to estimate the appropriate tax and freight charges, installation costs, and the indirect costs associated with the condensate system.

7.1.2.6 Yearly Incremental Capital Cost Bases. For those landfills that are not closed and still accepting refuse, an additional capital investment will be required each year to collect the gas produced by the new refuse. The incremental amount of capital required to install additional extraction wells, well connectors, header pipe, and condensate pipe is equal to the ratio of the refuse acceptance rate to the refuse capacity. Once the direct capital cost has been determined for each item, the appropriate indirect cost factors are applied to the purchase price and subsequently added to the direct capital cost to estimate the incremental capital cost. This incremental cost will be incurred each year until the landfill reaches design capacity. All prices are updated to August, 1987 values using the Chemical Engineering[47] indices.

7.1.2.7 Replacement Costs. At the end of the first gas mover system's functional life, a new system will be sized for the maximum landfill gas flow rate expected in the next 15 years (the estimated system functional life). Replacement equipment will be sized every 15 years over the control period. Wells, well connections, header piping, and condensate system are not replaced.

7.1.3 Operating Cost Bases

The bases for the annual operating cost of the gas collection system are presented in Table 7-7 (1987 dollars). The operating costs are the yearly expenditures necessary to operate and maintain the gas collection system. These costs include operating and maintenance labor, operating materials, replacement parts, utility for the blower system only, and waste disposal. The indirect operating expenses include plant overhead, property, insurance, taxes, administration, and other costs associated with owning the equipment. It is assumed that it requires one full time operator to operate and maintain the gas collection system during the day.[50] An automatic control system is assumed to operate and control the gas collection system at night. It is also assumed that the computer maintaining the control system will shutdown the collection system and notify the facility's off-duty operator via a dial-up system in case of a malfunction. The condensate is adjusted for pH and disposed to a POTW.

7.2 DEVELOPMENT OF CONTROL SYSTEM COSTS

This section presents the capital and annual operating costs associated with the flare system and gas turbine system. The costs for both control options are based on systems designed to handle the maximum landfill gas production expected during the functional life of the equipment.

7.2.1 Bases For Flare System

As discussed in Chapter 4, the domestic municipal solid waste landfills that flare landfill gas use an enclosed ground flare. The primary components of the landfill flare system costed for this analysis are itemized in Table 7-8. The flare system consists of an enclosed flare with an automatic air damper for emission control to ensure the flare exhaust is smokeless. It is assumed that the landfill flare system can achieve a 98 percent volatile organic destruction efficiency without requiring additional combustion fuel such as natural gas. As mentioned in Section 4.2.1.2, combustion efficiencies greater than 98 percent are observed when methane concentrations are greater than 10 percent. A typical methane concentration in landfill gas is 50 percent. The flare is also assumed to operate 8760 hours per year.

TABLE 7-7. ANNUALIZED COST BASES FOR THE GAS EXTRACTION SYSTEM

Direct operating cost	Cost factor
1) Operating Labor	
a) Operator	8 man-hours/day, 365 day/year @ 7.42/hr[a]
b) Supervisor	15% of 1a[b]
2) Operating Material	Nominal[c]
3) Maintenance	
a) Labor	0.5 hr/shift, 1 shift/day, 365 day/year @ 8.16/hr[b]
b) Material	100% of 3a
4) Replacement Parts[c]	
5) Utilities	
a) Electricity, blower only	$0.0511/kwh[d]
6) Condensate Disposal	$.033/gallon condensate[e]
<u>Indirect Operating Costs</u>	
7) Overhead	80% of 1a + 1b + 3a[b]
8) Property, Insurance, Taxes, Administration	40% of Capital Costs[b]

[a]USDL, mill worker rate of 6.18 plus fringes of 20 percent; 1983.

[b]Reference 51, p. 3-12.

[c]Reference 52, p. 6-24.

[d]Reference 53.

[e]Reference 54.

TABLE 7-8. FLARE SYSTEM COMPONENTS

Flare Tip
Flare Pilots with Flame Safeguard
Flare Stack
Ignition Panel
Pipe Racks
Flare Guy Wires Support
Knockout Drums with Seals
Platforms
Manual or Automatic Dampers
Temperature Sensor
Temperature Controller
Flame Arrestor and Motor Operated Shut Off Valve

NOTE: References indicate that flare service (i.e., steam and air assistance) is not required for typical landfill flare systems.

Source: References 55, 56, 57, and 58.

Two empirically derived equations were used to calculate the installed capital and annual operating cost of the flare system. These equations were developed based on flare purchase costs provided by flare vendors and direct and indirect cost escalation factors from available literature.[59-62] This approach was taken in lieu of estimating the cost individual flare system components. The equations are presented in Figure 7-5, with supporting data in Table 7-9.

7.2.1.1 Flare System Capital Cost. Equation 1 in Figure 7-5 is used to estimate the total installed capital cost of the flare system as a function of the input gas flow rate. The capital cost for the entire flare control system includes the purchase and installation of all equipment, pipe or duct, and pipe supports. The exhaust from the gas moving system (part of the gas extraction system is assumed to transport the landfill gas to the flare system. Therefore, a gas moving system is not included in the flare system.

7.2.1.2 Flare System Operating Costs. Equation 2 in Figure 7-5 is used to estimate the direct annual operating cost of the flare system. It is assumed in the derivation of Equation 2 that the direct annual operating cost equals 6 percent of the total installed capital costs.[63] An indirect operating cost equal to four percent of the total capital investment is added to the direct annual cost.

7.2.1.3 Flare System Replacement Costs. At the end of the functional life of the flare system, a new system is designed to handle the maximum landfill gas flow rate expected in the next 15 year equipment life. Replacement equipment will be sized and costed every 15 years during the control period.

7.2.2 Bases For Gas Turbine System

The gas turbine system cost is based on a simple-cycle, heat engine that converts the landfill gas, containing 50 percent methane, to electrical energy. More than 20 turbines are in use at 18 municipal solid waste landfills to recover energy from landfill gas.[64] It is assumed that the electrical energy produced by the turbine system is sold to a local utility. A recovery credit is incorporated in the annual cost of the turbine system to reflect electricity sold. Table 8-5 in Chapter 8 lists the electricity

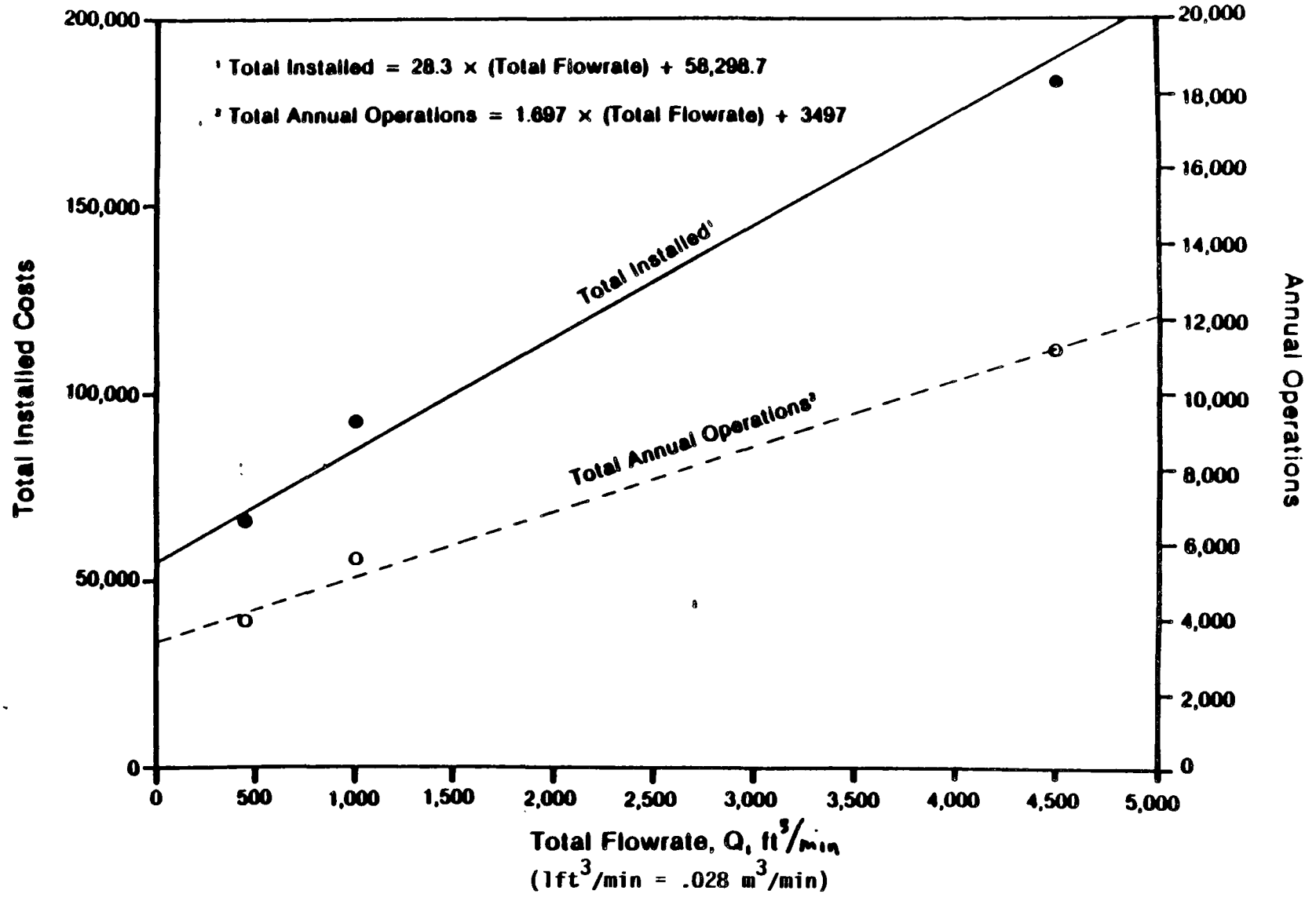

Figure 7-5. Flare equation for capital and annual operating costs.

TABLE 7-9. FLARE-BASES FOR TOTAL CAPITAL AND ANNUAL OPERATIONS COST

Capital Costs

I. Direct Capital Costs:

- Purchase Price, A

Flowrate, Q, ft^3/min[a]	Purchase Price, $[b]
450 ft^3/min	$25,000
1000 ft^3/min	35,000
4500 ft^3/min	70,000

- Installation

	Cost Factor[c]
- Foundation	6% of A
- Structure	15% of A
- Equipment Erection	15% of A
- Electrical	15% of A

- Total Base Cost, B, = A + .51A

- Sales Tax	25% of A + 25% of B
- Freight	16% of A
- Contractors Fees	30% of (B A)

II. Indirect Capital Cost:

- Total Contract, C, = B + Sales Tax, Freight, Fees

- Engineering	10% of C
- Contingencies	15% of C

Total Capital Cost = C + Engineering + Contingency

I. Direct Operating Costs:

Flowrate, Q, ft^3/min	Purchase Price, $[b]
450	$ 3,960
1000	5,544
4500	11,088

(continued)

TABLE 7-9. (Continued)

II. Indirect Operating Costs:[d]

Total Indirect Costs = 4% that of the total capital costs

[a]$1 ft^3$ min = .028 m^3/min.

[b]References 65 and 66.

[c]References 67.

[d]The annual operating cost is assumed to equal 6 percent that of the total installed capital costs. Organic Chemical Manufacturing, Volume 4: "Combustion Control Devices."

rates used by state. Section 7.2.2.1 presents the approach used to size the gas turbine system. Land requirements and an electrical switch gear station are included in the cost. The bases and method used to develop the total capital and annual cost are presented in Sections 7.2.2.2 and 7.2.2.3, respectively.

7.2.2.1 Gas Turbine Sizing. The size of the gas turbine system is based on the potential electrical output generated by using the landfill gas as fuel. The heat content of the landfill gas, based on a gas composition of 50 percent methane, is assumed to be 500 Btu/ft . The gas turbine system is considered to be 30 percent efficient in converting the landfill gas to electrical energy. It is recognized that a gas turbine with a power output of 2.93 to 29.3 MW will be subject to NO_x emission limits of 150 ppmvd at 15 percent oxygen.[68] However, it is assumed that this limit will be achievable with dry control technologies (i.e., combustion modifications). Therefore, the gas turbine system does not require wet controls to meet the NO_x emission limit. As with the gas collection system, the gas turbine is assumed to operate 8760 hours per year obtaining all electrical service from its own electrical generation.

7.2.2.2 Bases For Gas Turbine System Capital Cost. An empirically derived equation was used to calculate the installed capital cost of a simple-cycle, gas turbine and related equipment. This equation is based on reported costs for actual gas turbine installations. The capital cost for the gas turbine is shown in Table 7-10. The data are used to directly derive the net installed capital cost for gas turbines rather than to calculate and sum the capital cost of each individual component. Table 7-11 shows the equations used to estimate the capital cost associated with land requirements, an electrical switch gear system, and working capital.

Since most of the installed plant capital cost data are for cogeneration plants, this data is plotted against gas turbine output and is fitted to these points as shown in Figure 7-6. However, it is difficult to perform regression with only a few points for simple-cycle plants. Since the gas turbine is the major component of the plant cost for both types of plants, it is assumed that the line for simple-cycle turbines should have a slope similar to that of a cogeneration plant. Therefore, the line for

TABLE 7-10. INSTALLED CAPITAL COSTS FOR TURBINE PLANTS[a]

Type plant	Size (MW)	Cost (10^6 \$) 1983	Cost (10^6 \$) 1987	Source
Cogeneration	3.8	3.7	4.04	Turbomachinery, Ap. 83
Simple-cycle	3.1	1.25	1.36	Turbomachinery, Ap. 83
Simple-cycle	0.8	0.86	.94	Turbomachinery International Utility Costs Study, 1982
Cogeneration	20	11	12	Cogen. World, Summer 83
Cogeneration	19.6	6.7	7.3	GTW, Jan 83
Cogeneration	45	30	32.7	GTW, March-Apr 83
Cogeneration	75	25	27.3	GTW, Sept-Oct 83
Simple-cycle	50	25	27.3	GTW, Handbook, 79-80
Simple-cycle	63	17	18.6	Trip Report to El Paso Electric Company
Cogeneration	2.8	1.8	1.9	Amer. McG - 114 Response
Cogeneration	20	16	17.5	Cogen. World, Summer 83
Cogeneration	0.8	2.2	2.4	GTW Sept-Oct 83
Cogeneration	1.1	2.5	2.7	Cogen. World, Summer 83
Cogeneration	0.65	1.5	1.6	Cogen. World, Summer 83

[a]Data shown are from References 69 through 77.

[b]All costs corrected to 1987 dollars using the CE plant cost index 344.9.

TABLE 7-11. EQUATIONS USED TO COST ANALYZE THE GAS TURBINE

A. Simple-cycle plant capital costs[a]

$$(10^6\ \$) = 0.84\ (TMW)^{0.7} \quad (1)$$

where,

TMW = total electrical output, MW

B. Land costs[b]

$$(10^3\ \$) = AA\ (21{,}961)/1000 \quad (2)$$

where,

AA = acres required

C. Switch Gear[c]

$$\$ = 85{,}000 \left(\frac{MW}{2.5}\right)^{.6} \quad (3)$$

where,

MW = electrical output

D. Working Capital[d]

$$\$ = 25\%\ \text{of (direct operating costs)} \quad (4)$$

E. Operating Labor

$$(10^3\ \$/\text{year}) = (DLC \times HRS)/1000 \quad (5)$$

where,

DLC = $18.64/hr
HRS = hours per year worked

(continued)

TABLE 7-11. (Continued)

F. Supervisory Labor

10^3 \$/year) = SLC x .15 x HRS)/1000 (6)

where,

SLC = annual supervisory labor hourly charge rate equal to \$24.24/hr
HRS = hours per year an operator works

G. Maintenance Costs[e]

(10^3 \$) = .00275 (TMW) (HRS) for < 10 MW (7)

or

(10^3 \$) = .00125 (TMW) (HRS) for $\leq$ 10 MW (8)

TMW = total electrical output, MW
HRS = hours per year of operation

H. Payroll Overhead

\$/year = 30% of (operating labor + .5 [maintenance cost] + supervisory labor) (9)

I. Plant Overhead

\$/year = 26% of (operating labor + supervisory labor) (10)

J. G & A, Taxes, Insurance

\$/year = 4% of (replacement capital cost of turbine system) (11)

K. Interest on Working Capital

\$/year = 10% of (working capital + land) (12)

(continued)

TABLE 7-11. (Continued)

Cost Bases		
Cost item	Unit	Cost factor
Operating labor[f]	$/hr	$18.64
Supervision[g]	$/hr	$24.24
Land costs	$/acre	$21,961
Fuel electricity	$/kw-hr	0

[a]Simple-cycle plant capital costs are based on plant cost data obtained from gas turbine user and literature sources. References 78, 79, and 80 through 86.

[b]A price of $21, 961 per acre is an assumed in this cost analyses.

[c]References 87.

[d]The working capital is assumed to be 25 percent of the direct operating cost.

[e]Includes both maintenance labor, and materials. References 88 and 89.

[f]Industrial boiler cost report, August 31, 1982, Table 2-11 escalated to 1987 dollars using CE index.

[g]A 30 percent premium above operating labor.

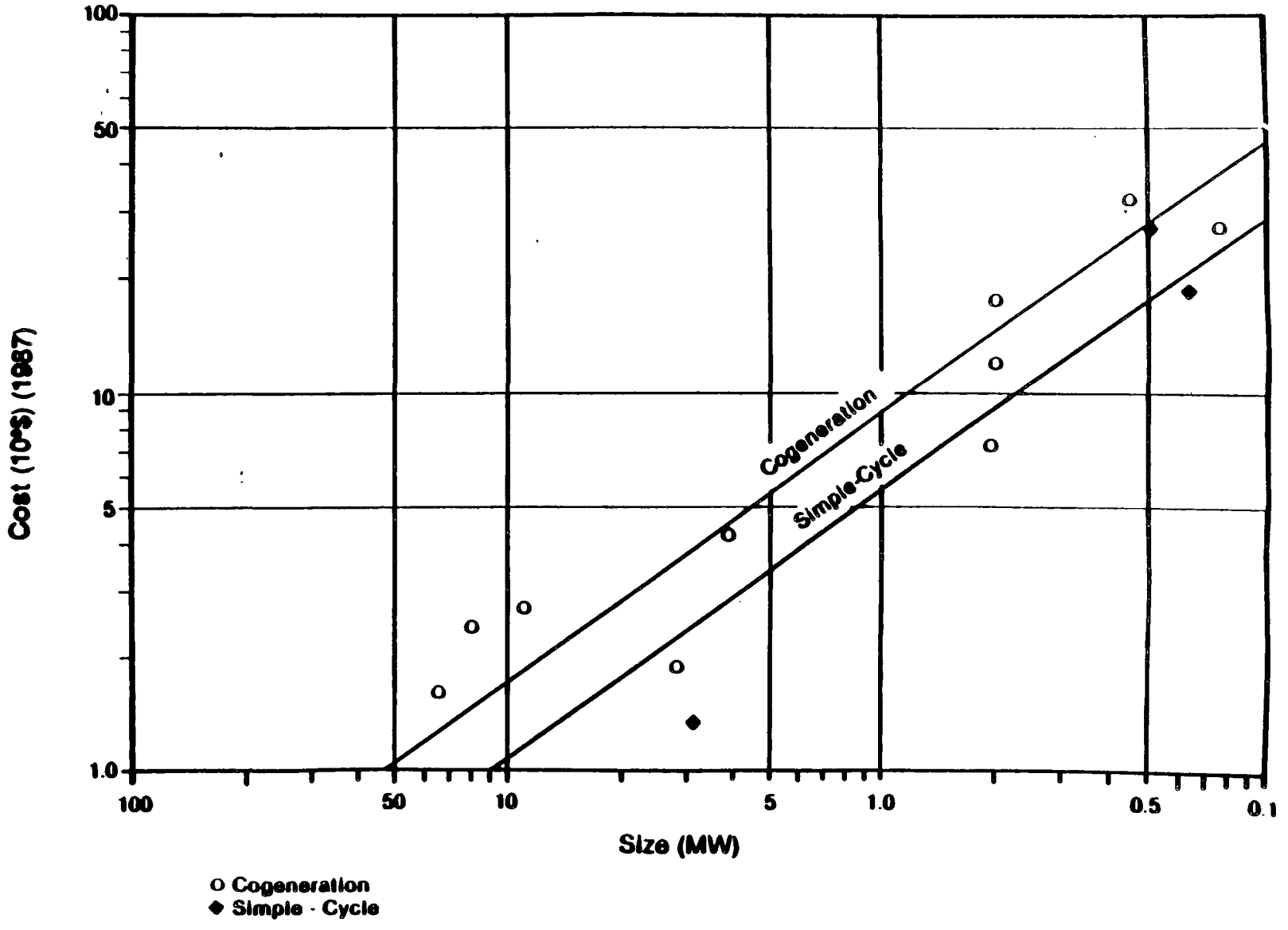

Figure 7-6. Turbine capital cost.

simple-cycle turbines is based on the available data, but drawn to the line for cogeneration. These lines and their respective equations are shown in Figure 7-6 and represent 1983 dollars.

Based on several plant visits, the average amount of land needed for a simple-cycle gas turbine is between 1 and 1.5 acres. The amount of land is broken down into turbines under 10 MW in size and turbines over 10 MW in size. Equation 2 in Table 7-11 is used to estimate the capital cost for land based on the acreage required.

Land prices will be a very small percentage of the total capital costs. In some cases, there would not be any land costs associated with the gas turbine. This would be the case if the gas turbine and equipment are to be located in an area already owned by the landfill facility. Therefore, land capital costs are conservatively included in the total turbine capital cost.

Equation 3 in Table 7-11 was used to estimate the capital cost associated with the electrical switch gear. Equation 3 is derived from electrical switch gear cost information reported in a Section 114 response and by applying the "six-tenths-factor" rule.[90] It is assumed with the use of this equation that the capital cost of the switch gear is a direct function of the gas turbine output.

The final item included in the initial capital investment of the turbine system is the working capital. This is assumed to be 25 percent of the direct operating cost.

7.2.2.3 Operating Costs For the Gas Turbine System. The components and operating cost bases are presented in Table 7-11. This section provides the bases for estimating direct operating costs. Included in the direct operating costs are operating and supervising labor, maintenance labor, and maintenance materials. The bases for estimating indirect operating costs are not discussed because these costs are simply estimated with the factors shown in Table 7-11.

- Direct Operating Costs[91-95]

  - Operating and supervising labor. Data on the operating labor requirements for the gas turbines are shown in Table 7-12. Based on these data, gas turbines require one operator whenever the turbine is operating. However, this assumption is probably conservative. For example, simple-cycle turbines (less than 20 MW) would likely not require a full-time operator. Therefore, in this cost analysis, it is assumed that only one operator will be required during the day time hours. The gas turbines will be operated by an automatic controller during the night time hours.

  - The capital cost associated with supervising labor is assumed to be equal to 15 percent of the operating labor plus an additional 30 percent salary premium.

- Maintenance Labor and Materials[96,97]

  Comments received from Solar Gas Turbines, Inc., and Dow Chemical regarding typical maintenance labor and material costs are used in the development of Equations 7 and 8 in Table 7-11.[96,97] Dow's comments indicated that total maintenance costs are $.002/KWH for aircraft derived gas turbines. Solar estimated that total maintenance costs for small turbines range from $.002/KWH to .0035/KWH. The hours of operation is assumed to be 8760 per year.

7.2.2.3 Gas Turbine Replacement. At the end of the first gas turbine's equipment life, a new system will be sized and costed to accommodate the maximum expected during the next 15 year equipment life. Replacement costs will be estimated for every 15 year interval in the control period.

7.2.3 Cost Effectiveness

The cost effectiveness of the flare and gas turbine options is estimated using two different economic approaches: single stage discounting and two-stage discounting. Single stage discounting is used to reflect the impact to industry, while two stage discounting is used to reflect the impact to society.

With single stage discounting, the cost effectiveness is calculated by dividing the net present value of the costs by the net present value of the emission reduction. This method is equivalent to the conventional method of dividing the annual cost by the annual emission reduction since both the

TABLE 7-12. DATA ON OPERATING PERSONNEL REQUIREMENTS[a]

	Approximate turbine size MW	Number of gas turbines	Number of operators per shift	Operators per turbine	Application
Houston Lighting and Power Wharton Station	50	8	7	.9	Combined-cycle
El Paso Electric	60	2	2[b]	1	Combined-cycle
Crown Zellerbach	30	1	1[c]	1	Combined-cycle
Southern California Edison, Long Beach	50	7	5	0.7	Combined-cycle
Southern California	70	4	4	1	Combined-cycle

[a]References 98, 99, and 101.

[b]One operator plus a legman.

[c]There is one additional person in the control room to assist the gas turbine operator if necessary. However, this additional person mainly takes care of plant systems not connected with the gas turbine cogeneration system.

costs and the emission reductions occur over the same time period. In the single stage discounting of costs and emission reduction, interest rates of four and eight percent were used for publicly and privately owned landfills, respectively.

Two stage discounting is used in situations where the capital costs imposed by regulations are likely to be passed on directly through to consumers.[102] In this approach, the estimated capital costs of a regulation are first annualized from the year the cost is incurred to the year the equipment is removed using the marginal rate of return on capital. In this cost analysis, a 10 percent marginal rate of return is used. Both benefits and costs are then discounted at the social rate of time preference. In other words, the annualized capital costs, actual operating costs, and actual emission reductions are brought back to some reference year using a three percent social rate and then are annualized over the total control period using the same social rate of time preference. The cost effectiveness can then be calculated by dividing the net present value of the costs by the net present value of the emission reduction or by dividing the annualized cost by the annualized emission reduction.

7.3 CONTROL COSTS FOR MODEL LANDFILLS

This section provides a comparison of the costs associated with the control of landfill gas at three stringency levels: 25 Mg NMOC/yr, 100 Mg NMOC/yr and 250 Mg NMOC/yr. Two model landfills were selected from the OSW database of landfills to represent the typical cost of controlling landfill air emissions for new and existing landfills. The landfills were selected based on their size, age, and gas generation factors which are typical of the landfills in the database. The physical characteristics of these model landfills are provided in Table 7-13.

7.3.1 Capital and Operating Costs

Tables 7-14, 7-15, and 7-16 show the year-to-year control costs for a typical existing landfill at stringency levels of 250 Mg NMOC/yr, 100 Mg NMOC/yr, and 25 Mg NMOC/yr, respectively. Tables 7-17, 7-18, and 7-19 show the control costs for a new landfill at stringency levels of 250 Mg NMOC/yr, 100 Mg NMOC/yr, and 25 Mg NMOC/yr, respectively. Only the first 20 years of the control period are shown in these tables for simplicity. In many cases,

TABLE 7-13. MODEL LANDFILLS[a]

Landfill characteristic	Existing model	New model
Design capacity (Mg refuse)	6,986,160	5,949,500
Age in 1992 (years)	20	(open 1994)
Depth (feet)	250	25
Average acceptance rate (Mg refuse/yr)	253,405	297,475
NMOC concentration[b] (ppmv)	6,381	6,381
Methane generation rate constant[b] (1/yr)	.028	.008
Methane generation potential[b] (ft^3/Mg refuse)	6,350	8,120
Type of owner	Public	Public

[a]Information extracted from the EPA Survey of Municipal Solid Waste Landfills (Reference 103).

[b]These values were randomly assigned to the landfills in the EPA database. See Chapter 3 for further discussion of these variables.

TABLE 7-14. ESTIMATED CONTROL COSTS FOR THE EXISTING MODEL LANDFILL AT A STRINGENCY LEVEL OF 250 Mg NMOC/yr

FLARE AND EXTRACTION SYSTEM COST BREAKDOWN

YEAR OF CONTROL (a)	1	2	3	4	5	6	7	8	9	10	11	12	13	14	15	16	17	18	19	20
NMOC EMISSION (Mg/yr)	902	936	968	999	1030	1059	1088	1116	1144	1170	1138	1106	1076	1046	1017	989	962	935	910	884
REFUSE IN PLACE (Mg)	5068096	5321501	5574906	5828311	6081716	6335121	6588526	6841931	6986160	6986160	6986160	6986160	6986160	6986160	6986160	6986160	6986160	6986160	6986160	6986160
WELLS INSTALLED	23.8923	1.1946	1.1946	1.1946	1.1946	1.1946	1.1946	1.1946	0.6799	0	0	0	0	0	0	0	0	0	0	0
FLARE CAPITAL COST ($)	153306	0	0	0	0	0	0	0	0	0	0	0	0	0	0	138613	0	0	0	0
EXTRACTION CAPTIAL COST ($)	768062	35588	35573	35558	35544	35532	35520	35508	20205	0	0	0	0	0	0	35671	0	0	0	0
TOTAL CAPITAL COST	921368	35588	35573	35558	35544	35532	35520	35508	20205	0	0	0	0	0	0	174284	0	0	0	0
FLARE OPERATING COST ($)	0	14184	14341	14494	14643	14787	14928	15064	15197	15326	15169	15016	14867	14723	14582	14445	13725	13595	13470	13347
EXTRACTION OPERATING COST ($)	0	83381	84882	86380	87876	89370	90861	92349	93223	93287	93210	93135	93062	92991	92923	94283	94218	94155	94093	94033
TOTAL OPERATING COST ($)	0	97565	99223	100875	102519	104157	105788	107414	108421	108613	108379	108151	107929	107714	107505	108728	107942	107750	107563	107381
NMOC EMISSION REDUCTION (Mg/yr)	0	917	949	979	1009	1038	1067	1094	1121	1147	1115	1084	1054	1025	997	969	943	917	891	867

NPV OF CAPTIAL COST ($) (b)=	3,363,800
NPV OF OPERATING COST ($)=	2,982,444
NPV OF NMOC EMISSION REDUCTION (Mg)=	22,228
COST EFFECTIVENESS ($/Mg)=	286

(a) Under this stringency level, this landfill was controlled for 64 years.
(b) NPV = net present value. Costs and emissions were brought back to the base year, 1992, at a rate of 3 %.

TABLE 7-15. ESTIMATED CONTROL COSTS FOR THE EXISTING MODEL LANDFILL AT A STRINGENCY LEVEL OF 100 Mg NMOC/yr

FLARE AND EXTRACTION SYSTEM COST BREAKDOWN

YEAR OF CONTROL (a)	1	2	3	4	5	6	7	8	9	10	11	12	13	14	15	16	17	18	19	20
NMOC EMISSION (Mg/yr)	902	936	968	999	1030	1059	1088	1116	1144	1170	1138	1106	1076	1046	1017	989	962	935	910	884
REFUSE IN PLACE (Mg)	5068096	5321501	5574906	5828311	6081716	6335121	6588526	6841931	6986160	6986160	6986160	6986160	6986160	6986160	6986160	6986160	6986160	6986160	6986160	6986160
WELLS INSTALLED	23.8923	1.1946	1.1946	1.1946	1.1946	1.1946	1.1946	1.1946	0.6799	0	0	0	0	0	0	0	0	0	0	0
FLARE CAPITAL COST ($)	153306	0	0	0	0	0	0	0	0	0	0	0	0	0	0	138613	0	0	0	0
EXTRACTION CAPTIAL COST ($)	768062	35588	35573	35558	35544	35532	35520	35508	20205	0	0	0	0	0	0	35671	0	0	0	0
TOTAL CAPITAL COST	921368	35588	35573	35558	35544	35532	35520	35508	20205	0	0	0	0	0	0	174284	0	0	0	0
FLARE OPERATING COST ($)	0	14184	14341	14494	14643	14787	14928	15064	15197	15326	15169	15016	14867	14723	14582	14445	13725	13595	13470	13347
EXTRACTION OPERATING COST ($)	0	83381	84882	86380	87876	89370	90861	92349	93223	93287	93210	93135	93062	92991	92923	94283	94218	94155	94093	94033
TOTAL OPERATING COST ($)	0	97565	99223	100875	102519	104157	105788	107414	108421	108613	108379	108151	107929	107714	107505	108728	107942	107750	107563	107381
NMOC EMISSION REDUCTION (Mg/yr)	0	917	949	979	1009	1038	1067	1094	1121	1147	1115	1084	1054	1025	997	969	943	917	891	867

NPV OF CAPTIAL COST ($) (b)=	3,677,897
NPV OF OPERATING COST ($)=	3,298,747
NPV OF NMOC EMISSION REDUCTION (Mg)=	22,755
COST EFFECTIVENESS ($/Mg)=	307

(a) Under this stringency level, this landfill was controlled for 96 years.
(b) NPV = net present value. Costs and emissions were brought back to the base year, 1992, at a rate of 3 %.

TABLE 7-16. ESTIMATED CONTROL COSTS FOR THE EXISTING MODEL LANDFILL AT A STRINGENCY LEVEL OF 25 Mg NMOC/yr

FLARE AND EXTRACTION SYSTEM COST BREAKDOWN

YEAR OF CONTROL (a)	1	2	3	4	5	6	7	8	9	10	11	12	13	14	15	16	17	18	19	20
NMOC EMISSION (Mg/yr)	902	936	968	999	1030	1059	1088	1116	1144	1170	1138	1106	1076	1046	1017	989	962	935	910	884
REFUSE IN PLACE (Mg)	5068096	5321501	5574906	5828311	6081716	6335121	6588526	6841931	6986160	6986160	6986160	6986160	6986160	6986160	6986160	6986160	6986160	6986160	6986160	6986160
WELLS INSTALLED	23.8923	1.1946	1.1946	1.1946	1.1946	1.1946	1.1946	1.1946	0.6799	0	0	0	0	0	0	0	0	0	0	0
FLARE CAPITAL COST ($)	153306	0	0	0	0	0	0	0	0	0	0	0	0	0	0	138613	0	0	0	0
EXTRACTION CAPTIAL COST ($)	768062	35588	35573	35558	35544	35532	35520	35508	20205	0	0	0	0	0	0	35671	0	0	0	0
TOTAL CAPITAL COST	921368	35588	35573	35558	35544	35532	35520	35508	20205	0	0	0	0	0	0	174284	0	0	0	0
FLARE OPERATING COST ($)	0	14184	14341	14494	14643	14787	14928	15064	15197	15326	15169	15016	14867	14723	14582	14445	13725	13595	13470	13347
EXTRACTION OPERATING COST ($)	0	83381	84882	86380	87876	89370	90861	92349	93223	93287	93210	93135	93062	92991	92923	94283	94218	94155	94093	94033
TOTAL OPERATING COST ($)	0	97565	99223	100875	102519	104157	105788	107414	108421	108613	108379	108151	107929	107714	107505	108728	107942	107750	107563	107381
NMOC EMISSION REDUCTION (Mg/yr)	0	917	949	979	1009	1038	1067	1094	1121	1147	1115	1084	1054	1025	997	969	943	917	891	867

NPV OF CAPTIAL COST ($) (b)=	3,740,000
NPV OF OPERATING COST ($)=	3,360,000
NPV OF NMOC EMISSION REDUCTION (Mg)=	22,800
COST EFFECTIVENESS ($/Mg)=	311

(a) Under this stringency level, this landfill was controlled for 108 years.
(b) NPV = net present value. Costs and emissions were brought back to the base year, 1992, at a rate of 3 %.

TABLE 7-17. ESTIMATED CONTROL COSTS FOR THE NEW MODEL LANDFILL AT A STRINGENCY LEVEL OF 250 Mg NMOC/yr

FLARE AND EXTRACTION SYSTEM COST BREAKDOWN

YEAR OF CONTROL (a)	1	2	3	4	5	6	7	8	9	10	11	12	13	14	15	16	17	18	19	20
NMOC EMISSION (Mg/yr)	265	292	315	338	361	383	406	428	450	472	493	490	486	482	478	474	470	467	463	459
REFUSE IN PLACE (Mg)	3272225	3569700	3867175	4164650	4462125	4759600	5057075	5354550	5652025	5949500	5949500	5949500	5949500	5949500	5949500	5949500	5949500	5949500	5949500	5949500
WELLS INSTALLED	99.6804	9.0619	9.0619	9.0619	9.0619	9.0619	9.0619	9.0619	9.0619	9.0619	0	0	0	0	0	0	0	0	0	0
FLARE CAPITAL COST ($)	98364	0	0	0	0	0	0	0	0	0	0	0	0	0	0	96793	0	0	0	0
EXTRACTION CAPTIAL COST ($)	981445	85342	85217	85106	85008	84919	84838	84764	84696	84634	0	0	0	0	0	23855	0	0	0	0
TOTAL CAPITAL COST	1079809	85342	85217	85106	85008	84919	84838	84764	84696	84634	0	0	0	0	0	120648	0	0	0	0
FLARE OPERATING COST ($)	0	8854	8966	9078	9188	9298	9407	9515	9622	9729	9834	9815	9796	9777	9758	9740	9659	9640	9622	9604
EXTRACTION OPERATING COST ($)	0	92486	95959	99428	102893	106354	109811	113265	116716	120164	120224	120214	120203	120192	120181	121125	121115	121104	121094	121084
TOTAL OPERATING COST ($)	0	101339	104925	108506	112081	115652	119218	122780	126339	129893	130059	130028	129999	129969	129940	130865	130773	130745	130716	130688
NMOC EMISSION REDUCTION (Mg/yr)	0	286	309	331	354	376	398	419	441	462	484	480	476	472	468	465	461	457	454	450

NPV OF CAPTIAL COST ($) (b)=	3,770,993
NPV OF OPERATING COST ($)=	2,707,316
NPV OF NMOC EMISSION REDUCTION (Mg)=	8,351
COST EFFECTIVENESS ($/Mg)=	776

(a) Under this stringency level, this landfill was controlled for 95 years.
(b) NPV = net present value. Costs and emissions were brought back to the base year, 1992, at a rate of 3 %.

TABLE 7-18. ESTIMATED CONTROL COSTS FOR THE NEW MODEL LANDFILL AT A STRINGENCY LEVEL OF 100 Mg NMOC/yr

FLARE AND EXTRACTION SYSTEM COST BREAKDOWN

YEAR OF CONTROL (a)	1	2	3	4	5	6	7	8	9	10	11	12	13	14	15	16	17	18	19	20
NMOC EMISSION (Mg/yr)	101	125	150	174	198	222	245	269	292	315	338	361	383	406	428	450	472	493	490	486
REFUSE IN PLACE (Mg)	1189900	1487375	1784850	2082325	2379800	2677275	2974750	3272225	3569700	3867175	4164650	4462125	4759600	5057075	5354550	5652025	5949500	5949500	5949500	5949500
WELLS INSTALLED	36.2474	9.0619	9.0619	9.0619	9.0619	9.0619	9.0619	9.0619	9.0619	9.0619	9.0619	9.0619	9.0619	9.0619	9.0619	9.0619	9.0619	0	0	0
FLARE CAPITAL COST ($)	94832	0	0	0	0	0	0	0	0	0	0	0	0	0	0	98364	0	0	0	0
EXTRACTION CAPTIAL COST ($)	390378	87173	86705	86350	86069	85840	85648	85484	85342	85217	85106	85008	84919	84838	84764	109077	84634	0	0	0
TOTAL CAPITAL COST	485210	87173	86705	86350	86069	85840	85648	85484	85342	85217	85106	85008	84919	84838	84764	207441	84634	0	0	0
FLARE OPERATING COST ($)	0	7899	8018	8136	8253	8370	8485	8599	8712	8825	8936	9047	9157	9266	9374	9481	9729	9834	9815	9796
EXTRACTION OPERATING COST ($)	0	68454	71988	75509	79018	82518	86010	89495	92974	96447	99916	103381	106842	110299	113753	118180	121628	121688	121677	121666
TOTAL OPERATING COST ($)	0	76353	80007	83645	87272	90887	94494	98094	101686	105272	108853	112428	115999	119565	123127	127661	131356	131522	131492	131462
NMOC EMISSION REDUCTION (Mg/yr)	0	123	147	170	194	217	240	263	286	309	331	354	376	398	419	441	462	484	480	476

NPV OF CAPTIAL COST ($) (b)=	4,278,308
NPV OF OPERATING COST ($)=	3,240,861
NPV OF NMOC EMISSION REDUCTION (Mg)=	9,453
COST EFFECTIVENESS ($/Mg)=	795

(a) Under this stringency level, this landfill was controlled for 116 years.
(b) NPV = net present value. Costs and emissions were brought back to the base year, 1992, at a rate of 3 %.

TABLE 7-19. ESTIMATED CONTROL COSTS FOR THE NEW MODEL LANDFILL AT A STRINGENCY LEVEL OF 25 Mg NMOC/yr

FLARE AND EXTRACTION SYSTEM COST BREAKDOWN

YEAR OF CONTROL (a)	1	2	3	4	5	6	7	8	9	10	11	12	13	14	15	16	17	18	19	20
NMOC EMISSION (Mg/yr)	25	51	76	101	125	150	174	198	222	245	269	292	315	338	361	383	406	428	450	472
REFUSE IN PLACE (Mg)	297475	594950	892425	1189900	1487375	1784850	2082325	2379800	2677275	2974750	3272225	3569700	3867175	4164650	4462125	4759600	5057075	5354550	5652025	5949500
WELLS INSTALLED	9.0619	9.0619	9.0619	9.0619	9.0619	9.0619	9.0619	9.0619	9.0619	9.0619	9.0619	9.0619	9.0619	9.0619	9.0619	9.0619	9.0619	9.0619	9.0619	9.0619
FLARE CAPITAL COST ($)	89426	0	0	0	0	0	0	0	0	0	0	0	0	0	0	98364	0	0	0	0
EXTRACTION CAPTIAL COST ($)	126940	90857	88864	87832	87173	86705	86350	86069	85840	85648	85484	85342	85217	85106	85008	109282	84838	84764	84696	84634
TOTAL CAPITAL COST	216366	90857	88864	87832	87173	86705	86350	86069	85840	85648	85484	85342	85217	85106	85008	207646	84838	84764	84696	84634
FLARE OPERATING COST ($)	0	7321	7442	7563	7683	7802	7920	8037	8153	8268	8383	8496	8609	8720	8831	8941	9407	9515	9622	9729
EXTRACTION OPERATING COST ($)	0	57863	61484	65064	68618	72153	75674	79183	82682	86174	89659	93138	96612	100081	103546	107981	111438	114892	118343	121792
TOTAL OPERATING COST ($)	0	65183	68927	72628	76301	79955	83594	87220	90836	94443	98042	101634	105221	108801	112377	116922	120845	124408	127966	131520
NMOC EMISSION REDUCTION (Mg/yr)	0	50	74	98	123	147	170	194	217	240	263	286	309	331	354	376	398	419	441	462

NPV OF CAPTIAL COST ($) (b)=	4,377,648
NPV OF OPERATING COST ($)=	3,426,269
NPV OF NMOC EMISSION REDUCTION (Mg)=	9,644
COST EFFECTIVENESS ($/Mg)=	809

(a) Under this stringency level, this landfill was controlled for 119 years.
(b) NPV = net present value. Costs and emissions were brought back to the base year, 1992, at a rate of 3 %.

the control period will exceed 20 years. The control period for the model existing landfill at a stringency level of 250 Mg NMOC/yr is 64 years. At a stringency level of 25 Mg NMOC/yr, the control period is 108 years. For the new landfill, the control periods range from 95 to 119 years in going from the least stringent to the most stringent cutoffs.

As shown in the tables, a collection system and control device are installed when the emission rate exceeds the specified cutoff. The control device (in this case the control device is a flare) and some components of the collection system (such as the blower) are sized for the maximum expected landfill gas generation rate and installed in the first year of control. Extraction wells and required collection headers are also installed in the first year of control based on the existing refuse in place. As additional refuse is placed in the landfill, more extraction wells and headers are installed. As a result, capital costs are incurred when the landfills emissions reaches the cutoff and each year thereafter, until the landfill has reached capacity. After the landfill has reached its refuse capacity, capital costs are only incurred every 15 years to replace equipment.

It is assumed that the first year of control is spent installing the equipment and that operating costs are not incurred until the second year of control, as exemplified in Tables 7-14 through 7-19. The operating cost increases each year as refuse is accepted, until the landfill reaches capacity. Once the landfill reaches capacity, the operating cost becomes relatively constant until equipment must be replaced. Capital and operating cost estimates were developed using the methodologies and costs presented in Sections 7.1 and 7.2.

7.3.2 Cost Effectiveness

The two stage cost effectiveness of controlling NMOC emissions at the three stringency levels are also presented in Tables 7-14 through 7-16 for the existing landfill and Tables 7-17 through 7-19 for new landfills. The cost effectiveness for the existing landfill at a stringency level of 250 Mg NMOC/yr is approximately $290/Mg NMOC reduced. At the most stringent level, 25 Mg NMOC/yr, the cost effectiveness increases to $311/Mg NMOC. The

cost effectiveness for the new landfill ranges from $776/Mg NMOC to $809/Mg NMOC in going from the least stringent to the most stringent cutoffs.

The cost effectiveness values presented in Tables 7-14 through 7-19 are calculated from the capital and operating costs and the emission reduction incurred over the entire control period. The costs and emission reductions in each year are brought back to the net present value in 1992 at a rate of 3 percent as described in Section 7.2.3. The cost effectiveness is the total net present value of the control costs (capital plus operating) divided by the net present value of the emission reduction.

7.4 NATIONAL COST IMPACTS

This section presents the national cost impacts for both existing and new landfills for the stringency levels of 250 Mg NMOC/yr, 100 Mg NMOC/yr, and 25 Mg NMOC/yr. These national cost impacts were developed using the EPA survey of municipal solid waste landfills discussed in Chapter 3 and the cost estimation methods provided in Sections 7.1 and 7.2 The control costs were computed for each landfill in the survey datafile as shown in the model cases in Section 7.3. The costs were then scaled to the national level and summed to provide the national cost impact.

7.4.1 Existing Landfill Cost Impacts

The national cost impacts of controlling existing landfill air emissions at three stringency levels are presented in Table 7-20. At the least stringent cutoff, 250 Mg NMOC/yr, approximately 9.6×10^6 Mg NMOC (net present value) is reduced by controlling 386 landfills yielding an overall cost effectiveness of $407/Mg NMOC. At the most stringent level, 25 Mg NMOC/yr, the overall cost effectiveness is $927/Mg which results from reducing 1.3×10^7 Mg NMOC (net present value) from approximately 1,900 landfills.

7.4.2 New Landfill Cost Impacts

Table 7-21 presents the national cost impacts for new landfills at three stringency levels. At an overall cost effectiveness of $897/Mg NMOC, approximately 630,000 Mg NMOC (net present value) can be reduced from 41 landfills under the stringency level of 250 Mg NMOC/yr. At the stringency level of 25 Mg NMOC/yr, 247 landfills would be reducing

TABLE 7-20. NATIONAL COST IMPACTS OF CONTROLLING EXISTING LANDFILLS AT THREE STRINGENCY LEVELS

	Stringency Level (Mg NMOC/yr)		
	25	100	250
Total number of landfills affected	1,884	853	386
NPV capital cost (10^6\$)	6,440	4,830	2,400
NPV operating cost (10^6\$)	5,120	2,830	1,510
NPV NMOC emission reduction (10^6Mg NMOC)	12.6	11.2	9.6
Overall cost effectiveness (\$/Mg NMOC)	927	640	407

TABLE 7-21. NATIONAL COST IMPACTS OF CONTROLLING NEW LANDFILLS AT THREE STRINGENCY LEVELS

	Stringency Level (Mg NMOC/yr)		
	25	100	250
Total number of landfills affected	247	104	41
NPV capital cost (10^6\$)	788	548	362
NPV operating cost (10^6\$)	614	348	200
NPV NMOC emission reduction (10^6Mg NMOC)	.99	.83	.63
Overall cost effectiveness (\$/Mg NMOC)	1,416	1,081	897

approximately 990,000 Mg NMOC (net present value) at a cost effectiveness of $1,416/Mg NMOC. The overall nationwide cost effectiveness for the new landfills is slightly higher than the existing landfills because the NMOC emissions from the new landfills would not include NMOC's from co-disposal of hazardous waste as some of the existing landfills might.

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8. ECONOMIC IMPACTS

This chapter evaluates the economic impacts of the §111(d) Guidelines and §111(b) Standards under the Clean Air Act that EPA will propose for closed/existing and new landfills, respectively. Section 8.1 presents an overview of the management of municipal solid waste, including recycling, incineration, and landfilling alternatives. Section 8.2 provides a detailed profile of landfills. Section 8.3 briefly describes the regulatory alternatives and control options under consideration. It also discusses the implications of the assumptions underlying the economic analysis. Section 8.4 examines the main economic impacts of the relevant regulatory alternatives. Section 8.5 discusses emissions reductions and the cost-effectiveness of the regulatory alternatives. Section 8.6 analyzes some distributional impacts of the regulatory alternatives. Finally, Section 8.7 examines the sensitivity of the social costs of the regulatory alternatives to changes in the discount rate.

8.1 OVERVIEW OF MUNICIPAL SOLID WASTE MANAGEMENT

Figure 8-1 shows the flow of municipal solid waste (MSW) from generation to disposal. MSW is generated as a by-product of consumption and production. After collection, sorted and unsorted MSW is either directly landfilled, incinerated in a municipal waste combustor, or sent to a centralized recycling facility. Most residues from recycling and combustion are sent to sanitary landfills. The main exception is hazardous ash from combustors, which is sent to a hazardous-waste landfill.

Section 8.1.1 describes the sources and composition of MSW and discusses trends in waste generation. Section 8.1.2 discusses the collection transfer, and transportation of MSW. Section 8.1.3 discusses materials recovery through centralized recycling and source reduction. Finally, Section 8.1.4 examines the combustion and landfilling of MSW.

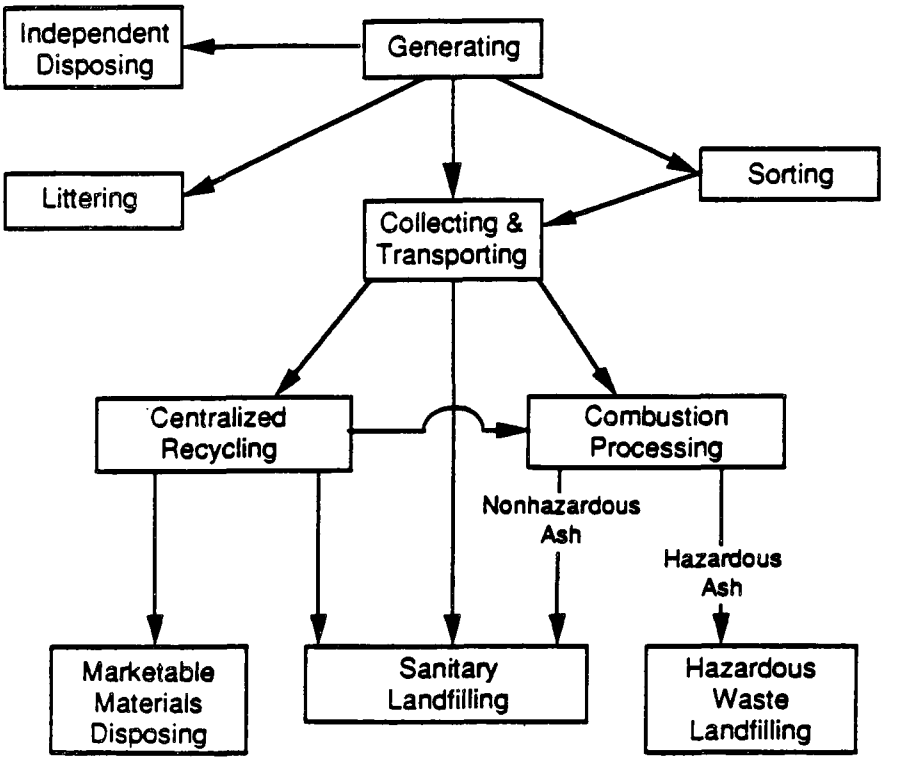

Figure 8-1. Flow of municipal solid waste from generation to disposal.[1]

8.1.1 Generation

Municipal solid waste includes all nonhazardous wastes from household, institutional, commercial, municipal and industrial sources.[2] Approximately 143 million Mg of MSW were generated in the United States in 1986.[3] This represents an average annual per capita generation of 0.60 Mg (about 3.6 pounds per day).[4] Table 8-1 presents the estimated quantities and shares of these discarded materials. Paper and paperboard comprise over 40% of gross discards. Yard wastes (e.g., grass clippings, tree trimmings, and leaves) represent the second largest portion - about 20%. Glass, metals, plastics, and food waste each comprise an additional 6 to 9% of the total.

As shown in Figure 8-2, generators of MSW can be classified into four broad groups:

- Residential,
- Commercial (e.g., offices, restaurants, and retail stores),
- Industrial (e.g., plants and factories), and
- Others.

The residential group generates approximately one-half of all municipal solid waste. The second largest group, commercial, generates about one-fourth of MSW. Most industrial by-products are either recycled, reused, or managed as hazardous wastes, leaving only a small portion to enter the municipal waste stream. Consequently, industrial sources are responsible for less than 5% of municipal solid waste. Other miscellaneous wastes such as sewage sludges and incinerator ash comprise about one-sixth of the municipal solid waste.

Various underlying factors influence the trends in the quantity of MSW generated over time. These factors include changes in population, individual purchasing power and disposal patterns, trends in product packaging, and technological changes that affect disposal habits and the nature of materials disposed.[7] Franklin Associates projects that MSW generation will increase at an annual rate of 1.4% over the period 1986 to 2000, and that about 175 million Mg of MSW will be generated in 2000.[8] This growth rate slightly exceeds estimates of population growth, reflecting an increase in annual per-capita generation from 0.60 to 0.73 Mg.[9]

TABLE 8-1. MATERIALS IN THE MUNICIPAL WASTE STREAM, 1986[5]

Materials	Quantity (10^6 Mg)	Share of gross discards (%)
Paper and paperboard	58.7	41.0
Glass	11.7	8.2
Metals	12.4	8.7
Plastics	9.3	6.5
Rubber and leather	3.6	2.5
Textiles	2.5	1.7
Wood	5.3	3.7
Food wastes	11.3	7.9
Yard wastes	25.7	18.0
Miscellaneous waste	2.4	1.7
Total wastes	143.0	100.0

These estimates exclude waste flows from demolition and construction, sludges, automobile bodies, nonhazardous industrial sources, incinerator residues, nonfood products discarded in containers, and packaging of imported goods. Details may not add to totals due to rounding.

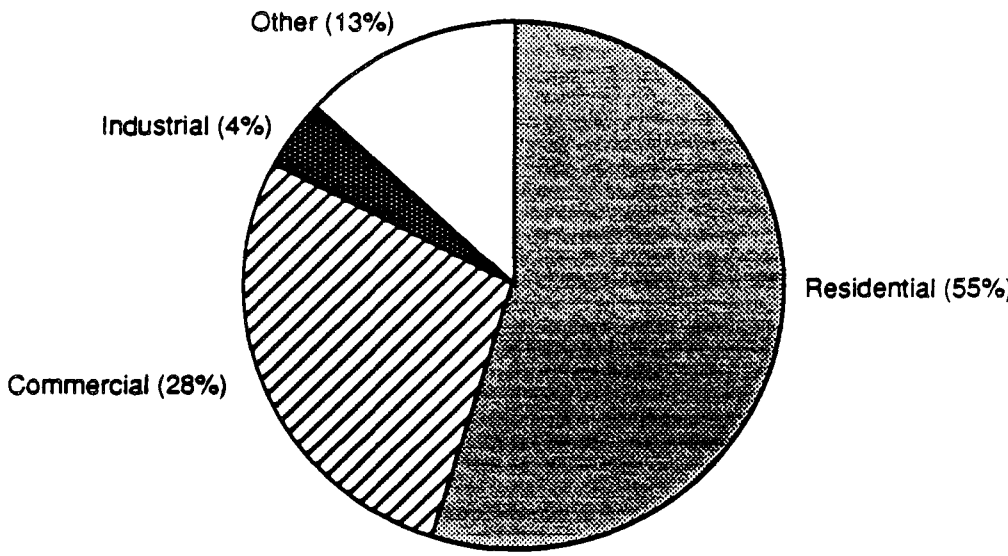

Figure 8-2. Sources of municipal solid waste.[6]

8.1.2 Collection and Transportation

Collection and transportation are necessary components of all MSW management systems regardless of the specific disposal options. MSW must be collected from generators and transported to a combustor, a recycling facility, a transfer station, or directly to a landfill. When recycling occurs, the residue must subsequently be transported to a combustor or landfill. Likewise, ash from municipal waste combustion must be transported to a landfill for disposal.

Collection of MSW varies by service arrangement between local governments and collectors and by level of service provided to households. The following five service arrangements account for over 99% of arrangements found in a 1978 National Science Foundation survey of municipalities:[10]

- Private - A private firm collects waste from households for a fee, but does not have exclusive territorial rights,
- Municipal - municipal employees collect waste,
- Contract - local government contracts with a private firm for the exclusive right to collect waste in a specified area. The private firm is paid by the local government,
- Self-service - households deliver waste directly to disposal sites or transfer stations, and
- Franchise - local government awards a private firm the exclusive right to collect waste in a specified area. The private firm collects fees directly from households.

The private and municipal service arrangements are the most popular with each used in about 30% of municipalities (see Figure 8-3). Contracts with private firms are found in 17% of municipalities, while 6% grant franchises. About 15% of municipalities collect waste under a self-service arrangement. Several different service areas may exist within a single municipality. For example, industrial and residential waste may be collected through different arrangements, or waste in different parts of a municipality may be collected by different contractors.

The level of service provided to households is usually linked to the frequency and location of pickup. With respect to collection frequency, about 60% of the cities collect waste once per week and 30% collect more

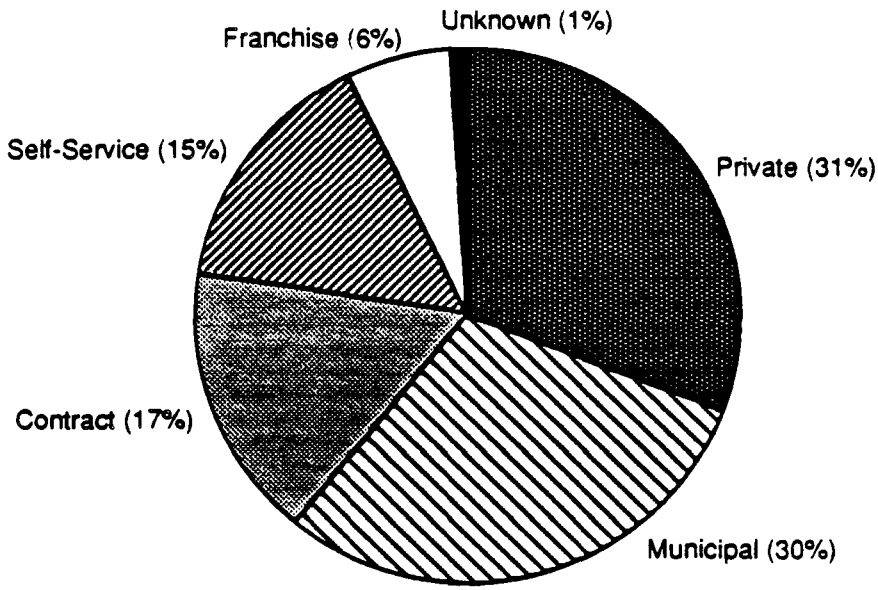

Figure 8-3. Service arrangements for MSW collection.[11]

frequently. The remaining 10% collect less frequently than weekly. The most common locations of pickup are backyard, curbside, and alley.[12]

Solid waste transfer is the process in which collection vehicles unload their wastes at centrally located transfer stations. Thus, smaller loads are consolidated into larger vehicles better suited for long-distance hauls. The larger vehicles then deliver the waste to the disposal site.[13] The larger vehicles are usually tractor-trailer trucks; however, trains and barges are becoming more popular for waste transfer.[14,15,16]

Transfer stations can decrease disposal costs in four basic ways: (1) hauling costs are decreased by decreasing the number of drivers and vehicles hauling waste to disposal sites, (2) turn-around time of collection vehicles is decreased when they do not have to haul waste to disposal sites, (3) larger vehicles can haul waste more efficiently allowing disposal at inexpensive distant disposal sites, and (4) some new transfer stations can recover marketable materials.[17,18,19]

As public opposition to MSW disposal facilities increases and the costs of disposal at locations near generators rise, long-distance hauls to disposal sites are becoming necessary.[20,21,22] Where very long haul distances are required, trains or barges are often used rather than tractor-trailers. Many waste planners prefer hauling waste by train or barge, because these modes of transportation are the safest and most invisible way to transport waste. They can also carry more weight legally and can be less expensive over very long hauls.[23]

8.1.3 Materials Recovery

As explained in EPA's Agenda for Action, the growing shortage of landfill space and the high cost of managing MSW make the recovery of materials from waste an attractive alternative to direct landfilling. Materials recovery increases the life of existing landfills by diverting potentially large quantities of waste from landfills.[24] Materials recovery also reduces the depletion of natural resources and removes toxic materials from the waste stream prior to disposal, enhancing the safety of landfilling and combustion.[25]

Materials are recovered from the municipal waste stream using two methods: source separation and centralized recycling. In the United

States, most materials are recovered through source separation, whereby waste generators manually separate materials for reuse or recycling before disposal. Disadvantages associated with source separation include both economic and perception problems. Existing waste collection vehicles are often ill-equipped and inefficient for curbside collection of separated materials. In addition, most people currently view discarded materials as "waste" rather than reusable materials. Thus, source separation programs often are not regarded as viable waste management options by decision-makers.[26]

Centralized recycling facilities separate marketable materials from the waste stream after collection. Transfer stations are often used in centralized recycling for sorting the waste for reusable materials during transfer.[27] The primary disadvantage of centralized waste processing is its high cost. Recycling activities are often not financially feasible when only the sale of recovered materials is considered. However, centralized recycling may be advantageous in communities where landfill disposal costs are high and there are substantial cost savings associated with reducing the size of the waste stream.[28]

8.1.4 Disposal Alternatives

After materials recovery, there are two options available for the management of collected MSW: landfilling and municipal waste combustion. Figure 8-4 presents a breakdown of 1986 gross MSW discards into landfilling, combustion, and materials recovery. As indicated, landfilling is the predominant MSW management option. In 1986, about 83% of gross MSW discards was landfilled and only 6% was incinerated. About 11% of gross discards was recycled.[30]

8.1.4.1 Landfilling. A landfill is an area of land or an excavation where waste is placed for permanent disposal. Municipal solid waste management uses two types of landfills: hazardous waste landfills and sanitary landfills. The primary purpose of hazardous waste landfills in the management of MSW is the disposal of hazardous ash residue from combustors. Sanitary landfills receive nonhazardous waste from residential, commercial and industrial sources and a small amount of small-quantity-generator hazardous waste.

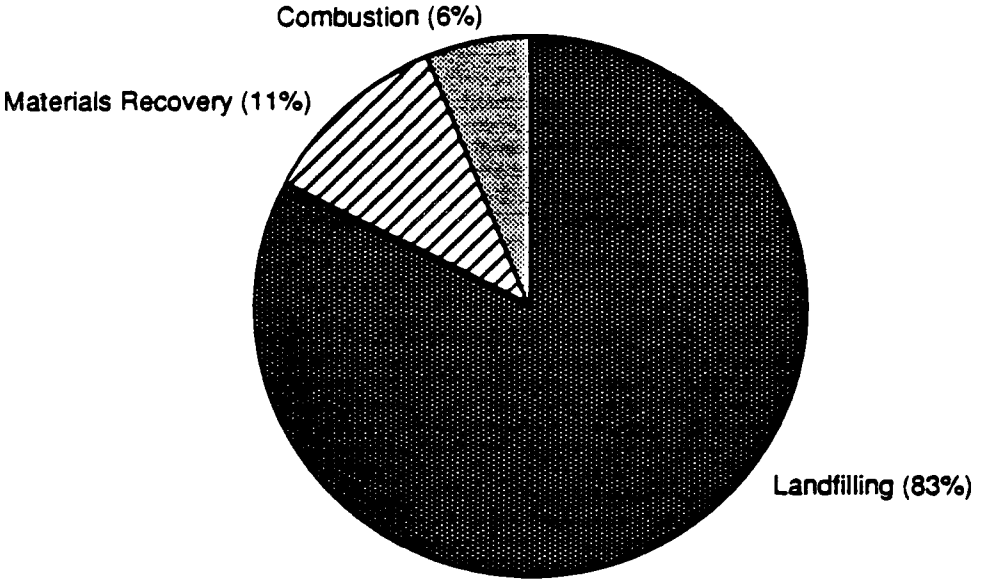

Figure 8-4. Share of MSW managed in disposal alternatives.[29]

Although landfilling is the predominant disposal alternative, it is becoming less attractive as an MSW management option. The public is becoming aware of the potential health, safety, and environmental impacts of landfilling (e.g., groundwater contamination, air emission of pollutants, and danger of explosion). As a result, public opposition toward landfilling has increased and a NIMBY (not-in-my-backyard) attitude toward landfills has become prevalent. Likewise, the regulatory environment surrounding landfilling is becoming increasingly stringent, making landfilling more expensive.[31] Increasing costs, public opposition, and land scarcity have closed many landfills and have made the siting of new landfills increasingly difficult.

The Office of Solid Waste in the U.S. Environmental Protection Agency projects that over 70% of existing landfills in 1986 will close by the year 2003, representing nearly three-fourths of the 1986 MSW landfill capacity. By 2013, only 10% of existing capacity in 1986 will remain. If current trends in landfill siting and development continue, only about one-third of lost capacity will be replaced.[32]

8.1.4.2 Municipal Waste Combustion. Municipal waste combustion (MWC) is the process of reducing the volume of MSW through incineration. MWC facilities range in design capacity from less than 25 Mg per day to more than 2000 Mg per day.[33] Combustion of MSW reduces waste volume by as much as 90%. Therefore, many municipal planners view MWC as an important method of reducing the need for additional landfill capacity.

Four broad categories of technologies are available for MWC: mass burn, modular, refuse-derived fuel (RDF), and other. In 1987, 161 MWC facilities with approximately 63,000 Mg per day of design capacity were in operation. Of existing capacity, approximately 55% was mass burn, 27% modular, 9% RDF, and 9% other.[34] See Figure 8-5.

Mass burn combustion requires no preprocessing of MSW other than the removal of very large items (e.g., tree trunks) and some mixing to produce a more homogeneous fuel.[36] Rams and/or grates move the waste through the combustor. Mass burn combustors can operate using either waterwall technology, which usually incorporates energy recovery, or refractory technology, which is an older, less efficient, design without energy recovery.[37]

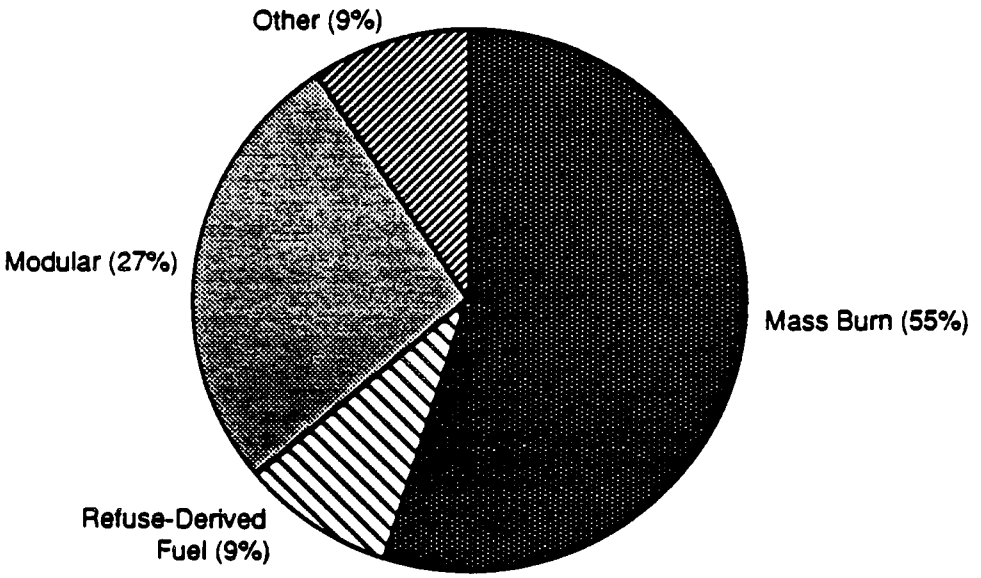

Figure 8-5. Municipal waste combustion technologies: Distribution of design capacity.[35]

Modular combustors consist of one or more factory prefabricated combustor units. Like mass burn facilities, modular combustors require minimal preprocessing of MSW and move waste through the combustor using either rams or grates. Modular combustor plants range in capacity from 25 to 500 Mg of MSW per day. Modular combustors incinerate waste using either a "starved air" design where the amount of oxygen is controlled to achieve pyrolysis or an "excess air" design where the amount of oxygen is not controlled.[38]

RDF combustors incinerate sorted and preprocessed MSW referred to as "refuse-derived fuel" (RDF). The sorting and preprocessing of MSW into RDF may or may not be performed at the same location as the combustor. Sorting is typically performed using a system of shredders, magnets, screens, air classifiers, and conveyers. Preprocessing of MSW ranges from simply removing noncombustibles and shredding to the production of high-quality fuel pellets. RDF yields a higher heat value, lower ash volume and more complete combustion than nonprocessed waste.[39]

Other MWC technologies include fluidized-bed gasification and fluidized-bed combustion. Combustors using fluidized-bed technologies incinerate MSW more efficiently than mass burn, modular, or RDF units by making the waste behave as a liquid or gas. However, fluidized-bed technologies are relatively new and still undergoing refinement.[40]

8.2 LANDFILL DISPOSAL OF MUNICIPAL SOLID WASTE

This section presents a profile of municipal solid waste (MSW) landfills. Section 8.2.1 describes some characteristics of municipal landfills. Section 8.2.2 discusses the costs of landfilling and methods of paying for landfill operations. Section 8.2.3 examines the changing regulatory environment in which landfills operate. Finally, Section 8.2.4 describes trends in landfilling MSW.

8.2.1 Characteristics of Municipal Solid Waste Landfills

Landfilling is defined as the disposal of waste through a three-step process that includes:[41]

- Spreading collected waste into thin layers in the landfill,
- Compacting the layers into the smallest practical volume, and
- Covering the compacted waste with soil on a daily basis.

EPA's National Survey of Solid Waste (Municipal) Landfill Facilities documented 6,034 landfills operating in 1986.[42] An estimated 535 landfills closed in 1987, leaving 5,499 landfills in operation in 1988.[43]

Landfills vary widely in the annual quantity of waste received, as Figure 8-6 shows. Most landfills receive small quantities of waste and relatively few receive very large quantities. The average annual quantity of waste received by landfills is 31,400 Mg, but an estimated 84% of landfills receive less than the average amount. The median amount of waste received, 2,570 Mg per year, better represents the typical landfill.[45]

Although landfills accepting over 100,000 Mg of MSW per year comprise only 8% of the landfill population, they manage an estimated 74% of landfilled waste. It is estimated that the largest 21 landfills (0.3%) receive over 23% of landfilled waste.[46] Landfills in the size category with the largest number of facilities, those accepting 907 to 9,070 Mg of MSW per year, receive less than 5% of landfilled waste.[47]

8.2.2 Technologies

Landfills generally use either the "trench" or "area fill" methods of landfilling, but combinations of the two methods are also used. Figure 8-7 presents the percentage of landfills using the trench or area fill methods of landfilling or a combination. About one-half of all landfills use the trench method exclusively, while about 30% of landfills use only the area fill method. Approximately 15% of landfills incorporate a combination of the trench and area fill methods. Only 5% of landfills use some other method.[49,50]

The trench method involves spreading and compacting the waste on the sloped end of an excavated trench. Cover material is obtained from the original trench excavation.[51] Trench landfilling has the following advantages:

- makes cover material readily available,
- exposes a minimum-size working face,
- gives optimum drainage during filling operations, and
- is easily adapted to wide variation in size of operation.

However, landfills using this method of landfilling must pay close attention to soil depth and groundwater conditions.

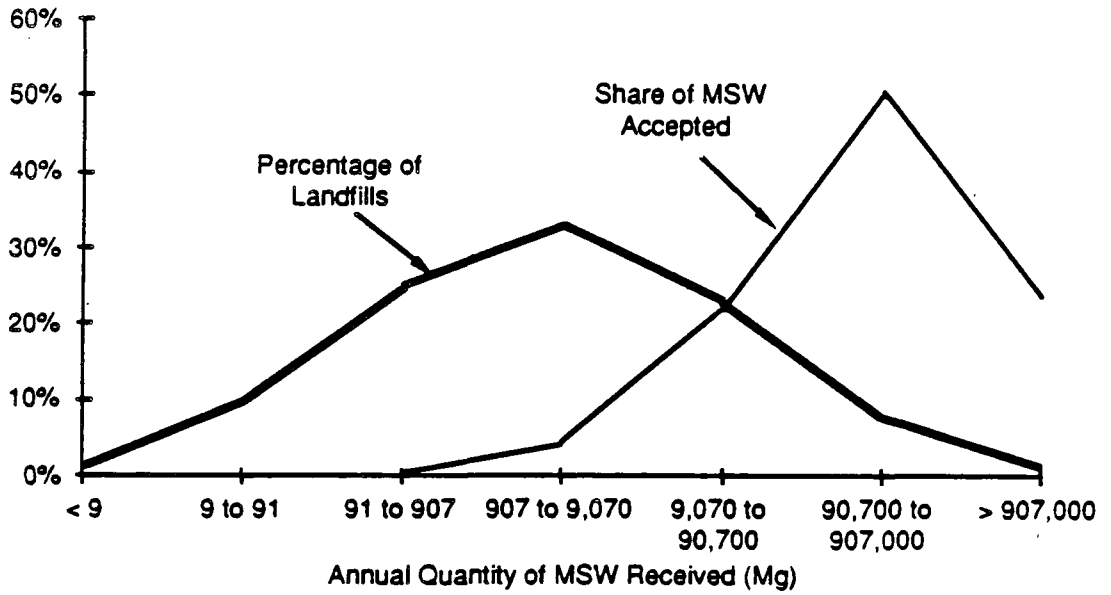

Figure 8-6. Distribution of annual quantity of MSW received at landfills.[44]

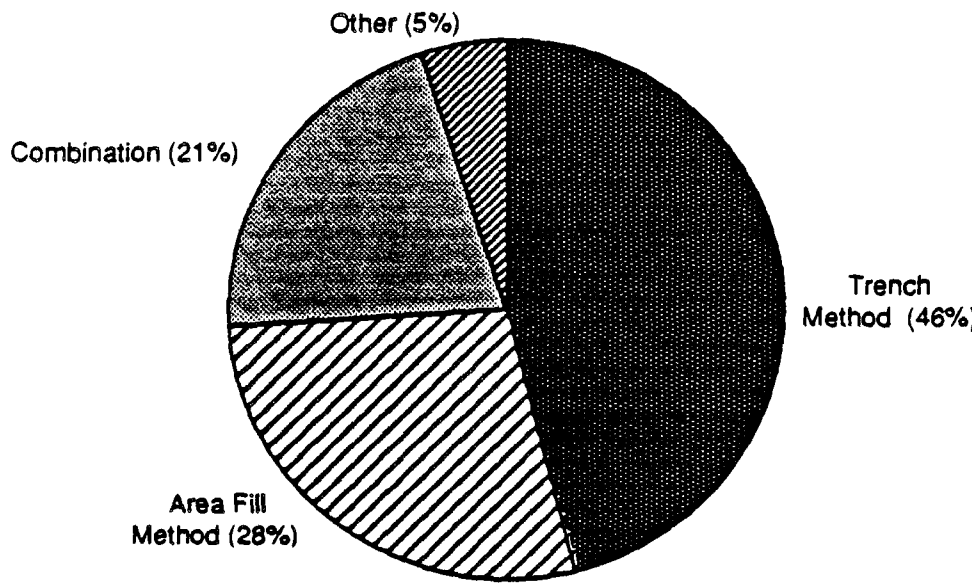

Figure 8-7. Landfill technologies.[48]

The area fill method involves spreading and compacting waste uniformly on the surface of the ground and covering with soil.[52] The cover material may be imported or may be excess material from trench landfilling. The area fill method is generally used when the land is gently sloping or land depressions are present.* Area fill landfilling accommodates large operations and is advantageous where groundwater conditions or soil depth do not allow excavation.[54] However, cover material is not always readily available, and drainage problems may require expensive liners.[55]

Landfilling methods combining the trench and area fill methods provide the flexibility to adapt site construction to the particular needs of a community. The "progressive slope" or "progressive ramp" method is one system where soil is excavated directly adjacent to the working face and spread over one day's waste. The remaining depression is then filled with the next day's waste, which is covered with soil from another adjacent excavation, and so on. The progressive slope method eliminates the need to import cover material, while allowing a portion of the discarded waste to be deposited below the original surface.

8.2.2.1 Environmental effects of landfills. The principal environmental concern when constructing and operating a landfill is the formation of highly contaminated leachate that can be discharged into surface water or groundwater. This leachate forms when precipitation or groundwater passes through the landfill or when water drains from discarded solid waste. EPA tests of leachate from municipal landfills have detected high concentrations of contamination by volatile organic chemicals (VOCs), acid organics, and base-neutral organics. Contamination by polychlorinated biphenyls (PCBs) and chlorinated pesticides was also found, but with less frequency.[56] The EPA estimated in 1987 that only 36% of landfills monitor groundwater near landfills and only 15% monitor for surface water contamination. However, only 2% of active landfills have ever been found to be a source of groundwater contamination.[57]

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*A variation of the area fill method is the "ramp" method. With the ramp method solid waste is spread and compacted on a slope. Cover material is obtained from directly in front of the working face and compacted on the waste.[53]

The most effective way to limit the harmful effect of leachate is to prevent its formation. Leachate generation can be limited by controlling the movement of water through the landfill cover and into the waste. Promoting runoff of precipitation and evaporation of water from cover material reduces the generation of leachate. This is usually accomplished by using soil with low permeabilities as cover material and increasing the slope of the landfill surface. Membrane or other nonsoil covers are used in areas where appropriate soil materials are unavailable or extraordinary environmental conditions exist.[58]

If hydrogeologic conditions indicate that leachate generation will cause harm to surrounding water resources, it becomes necessary to install a liner and possibly a leachate collection system. A properly designed liner will effectively limit the movement of leachate contaminants through the base of the landfill and into the underlying geologic formations. Liners may either physically prevent leachate movement or chemically remove contaminants from water that travels through the liner. At the same time the liner must withstand chemical and physical attacks from the decaying waste.

Liner composition may be of natural or synthetic materials. Common liner materials include:

- compacted soils and clays,
- admixes such as asphalt concrete or soil cement,
- polymeric membranes such as rubber and plastic sheetings,
- sprayed on linings,
- soil sealants, and
- chemisorptive liners.[59]

The EPA estimates that 60% of active landfills employ liners in leachate management. Approximately 87% of landfills with liners use a clay or soil material.[60]

Leachate usually accumulates in the bottom of lined landfills. If the leachate is not removed, pressure will build at low points in the liner possibly resulting in a discharge around the liner onto the ground surface. If the pressure builds to a very high level, the liner may become damaged and allow leachate to pass. To prevent such discharges, leachate collection systems hydraulically pipe leachate to the surface for treatment and

disposal. Treatment of leachate may include: recirculation back into the landfill, physical or chemical treatment, land disposal, or discharge to a sewer or surface water.[61] The EPA estimates that only 12% of active landfill units have leachate collection and treatment systems.[62]

A second environmental concern posed by landfills is the formation and release of methane gas. Decomposition of MSW begins immediately upon placement in a landfill. Initially, aerobic decomposition occurs and carbon dioxide gas is generated. This decomposition by bacteria begins after the supply of oxygen is depleted. This decomposition generates methane gas, which can continue for many years after landfill closure. The generation of methane gas at landfills is potentially dangerous because methane gas

- Is explosive in high concentrations,
- Can asphyxiate people and animals, and
- Kills vegetation as it passes through soil.[63]

Methane gas is recovered either with the use of gas recovery wells or passive venting systems. Methane gas may be burned off in flares immediately after collection, or the considerable energy content may be recovered. As discussed in Chapter 4, energy may be recovered from landfill gas in several ways, including:

- Upgrading gas to pipeline quality for delivery through natural gas distribution systems,
- Using gas as a boiler fuel to generate steam, and
- Generating electricity from the combustion of gas.[64]

The Public Utilities Regulatory Policies Act of 1978 (PURPA) provides significant incentives to recover energy from methane gas by requiring electric utilities to purchase electricity from small power producers such as landfills.[65] However, the EPA estimated in 1986 that only 7% of landfills monitored for methane gas and only 2% operated a gas recovery system.[66]

Proper closure of landfills is necessary once they are filled. Landfill closure typically involves installation of a final landfill cover or cap limiting the entry of water in order to control leachate generation.[67]

After closure, landfill sites have many potential uses ranging from golf courses to sites for commercial buildings. For example, Denver, Colorado's Mile High Stadium is constructed on a former landfill site.[68] However, long-term care of landfill sites may continue for an additional 20 to 30 years after closure under current conditions. Gas or leachate may migrate from the landfill if control mechanisms fail or were not installed. Monitoring for contamination and remedial action may be necessary. Similarly, inspections of the landfill cover and possible regrading to prevent ponding may be required.[69]

8.2.2.2 Ownership and Jurisdictions Served. Waste disposal sites, especially landfills, are likely to be owned and operated by public entities. Government institutions also play a large role in regulating the disposal of MSW. Local communities, in particular, often take the lead in MSW management. Many factors justify their interest, including concerns that: MSW may pose a threat to the public health, improperly disposed waste may result in adverse environmental impacts, and problems such as noise, traffic, and odor may result from the disposal of MSW. Municipal officials often believe that owning and operating landfills provide them with the necessary control over these factors.[70]

Figure 8-8 shows that over 85% of municipal landfills are publicly owned. The most common owners of landfill facilities are county and city governments, who together own nearly 60% of all landfills. The federal government owns 3% of existing landfills, which are mainly facilities on military installations. State governments own less than one percent of landfills. Less than 15% of landfills are owned by private entities.

Economies of scale exist in landfilling MSW, making the unit costs of operating small landfills relatively high compared to larger landfills. Consequently, it is usually not profitable for private waste disposal firms to operate small landfills. Figure 8-9 shows the ownership of landfills by size. Not surprisingly, large landfills are more likely to be privately owned than small landfills. About 32% of landfills receiving more than 180 thousand Mg of MSW per year are privately owned while nearly 90% of landfills receiving less than 900 Mg of waste per year are publicly owned.[73] Similarly, the median annual quantity of waste received is approximately

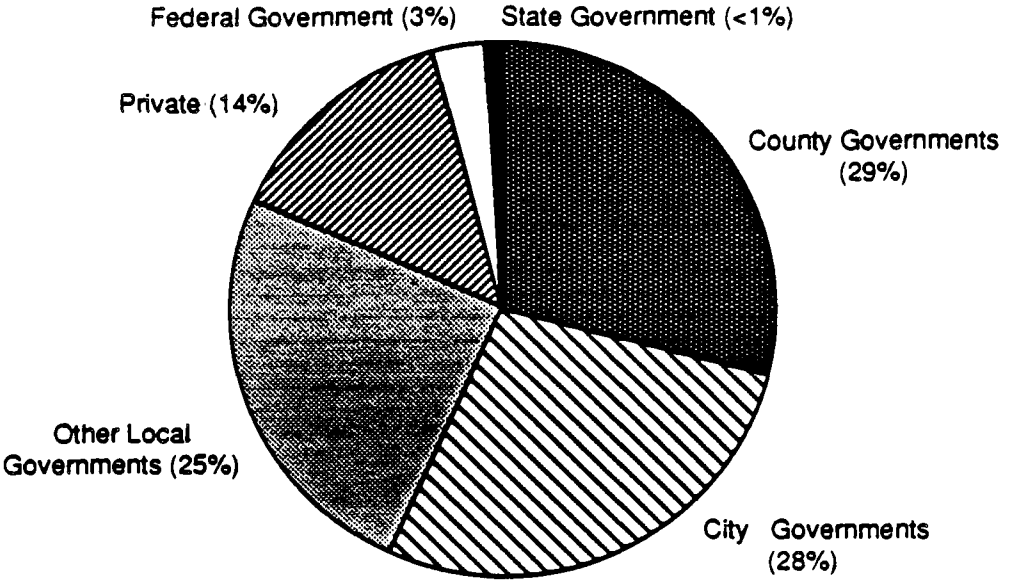

Figure 8-8. Ownership of municipal landfills.[71]

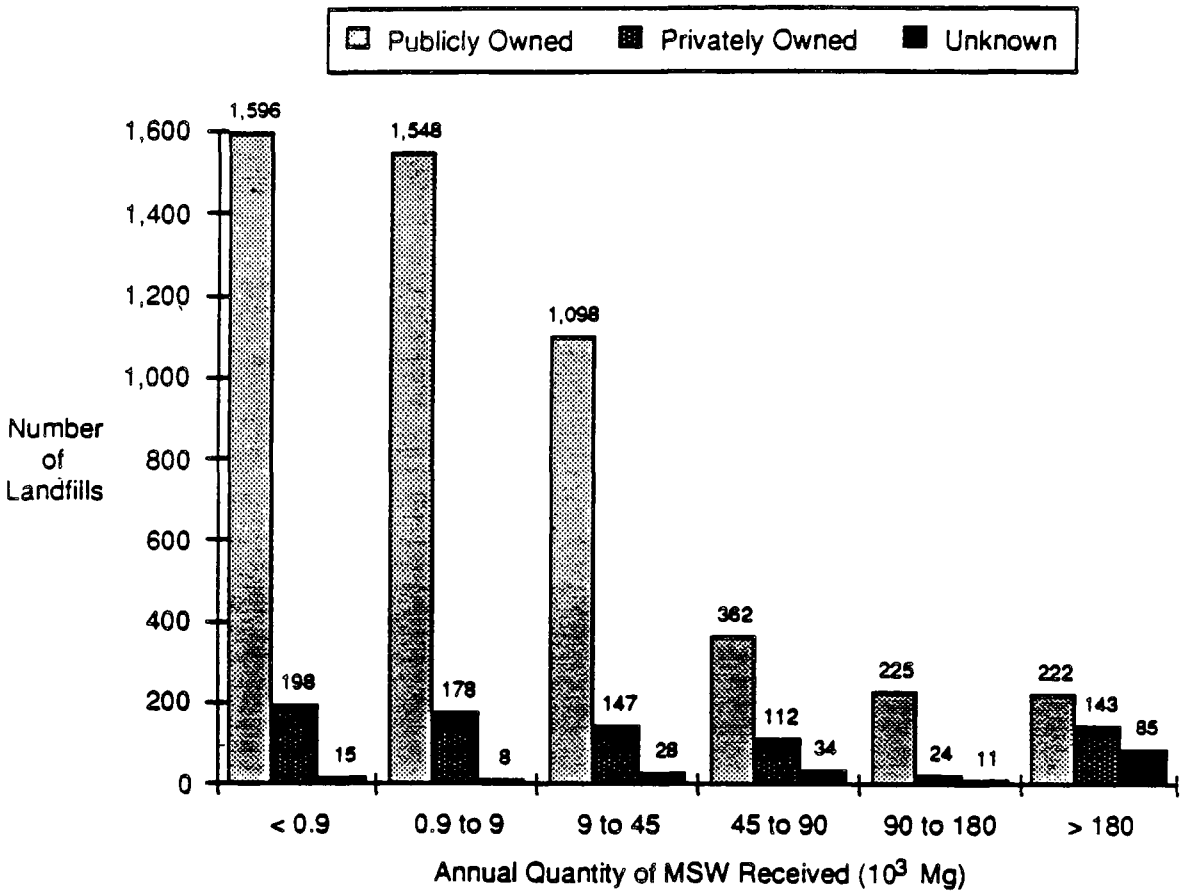

Figure 8-9. Ownership of landfills by size.[72]

7,000 Mg for publicly owned landfills and approximately 41,000 Mg for privately owned landfills.[74]

Landfills typically serve specific jurisdictions (see Figure 8-10). According to the EPA survey of municipal landfills about 77% of landfills serve specific jurisdictions. However, landfills may serve multiple jurisdictions. The actual number of jurisdictions served by landfills varies widely. While the average landfill serves only three jurisdictions, one landfill facility reported receiving waste from 53 specific jurisdictions.[76]

8.2.3 Costs of Municipal Solid Waste Landfilling

Landfill costs are frequently divided into five major categories:[77]

- pre-development,
- construction,
- operating,
- closure, and
- long-term care.

Pre-development costs include the costs associated with investigating available landfill sites and assessing their suitability. Pre-development costs generally represent less than 10% of total landfill site development costs and include expenditures associated with

- land acquisition,
- preliminary site engineering,
- preliminary legal services, and
- licensing and permit review.

Pre-development costs vary widely because of differences in land costs, state regulations, and the level of MSW management services desired. Land costs depend on the local real estate markets, the amount of land required, and the land's proximity to urban areas. As NIMBY attitudes toward landfills have increased, less expensive land farther from cities has been purchased. However, increasing transportation costs associated with higher fuel prices place limits on the distance that waste may reasonably be hauled. Similarly, engineering and legal costs have increased as state permitting processes have become increasingly complex.[78]

Landfill site construction costs include the major up-front expenditures and all construction costs throughout the life of the facility.

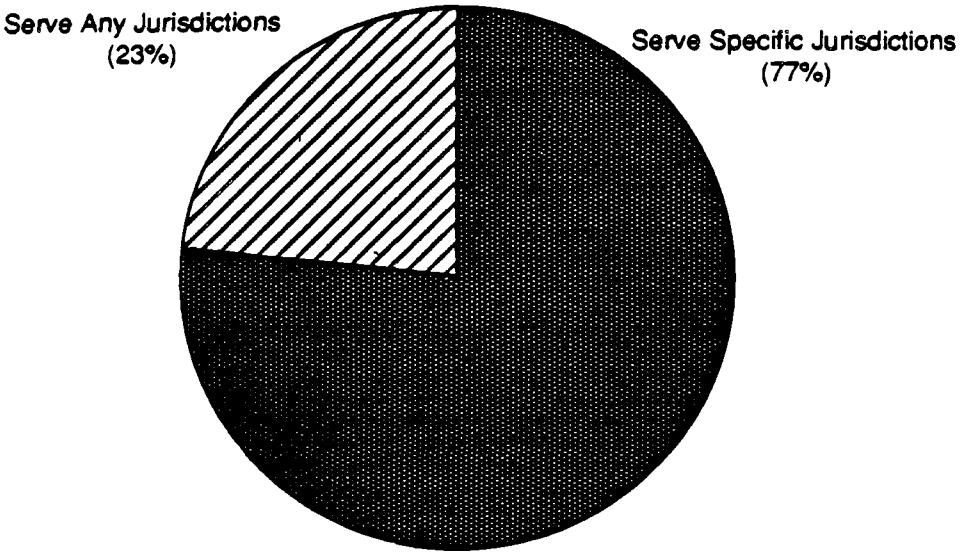

Figure 8-10. Jurisdiction limitations of municipal landfills.[75]

Landfill construction costs typically represent 25% to 35% of total landfill costs and include the costs of

- excavation and soil placement,
- liner construction,
- leachate collection system,
- surface water drainage controls,
- gas venting and/or collection system,
- facilities (buildings, etc.), and
- site-access (roads, etc.).[79]

Typically, costs associated with liner construction account for about 60% of construction costs. Therefore, a major factor in determining landfill construction costs is the type of liner design used. Other factors influencing construction costs include local economic conditions, haul distances for construction materials, and time of year.[80]

Operating costs represent the greatest portion of landfill costs. Operating costs typically represent 40% to 50% of total landfill costs and include expenditures associated with

- environmental monitoring (leachate, groundwater, surface water, landfill gas, and air emissions),
- maintenance,
- labor,
- utilities,
- administrative costs, and
- fuel for machinery.

Operating costs vary widely between landfills because of large differences in environmental monitoring. Landfills using "state-of-the-art" monitoring and collection systems are significantly more costly to operate than landfills incorporating older technologies.[81]

Closure costs typically represent the smallest share of total landfill costs. Depending upon the complexity of closure operations, costs usually range from 1% to 5% of total landfill costs. Closure costs include the costs of

- placing the final cover or cap on the landfill site,
- installing gas venting or collection systems, and
- documenting that the site has been properly closed.

Long-term care includes actions required by federal and state regulations to ensure that closed landfills present no danger to public health and safety. The costs of long-term landfill care can range from 10 to 15% of total landfill costs and include the costs of

- environmental monitoring,
- leachate treatment, and
- land surface care to ensure proper drainage of surface water.[82]

Landfill costs vary widely depending on the amount of waste disposed. Table 8-2 presents the unit costs of municipal solid waste landfills. These costs combine capital and operating costs into a single unit cost value. The costs suggest that significant economies of scale (unit costs decrease with increasing production) exist in landfilling MSW, as noted above. For example, MSW disposal costs $92.20/Mg of waste in a 10 Mg/day private landfill while disposal costs are only $10.60/Mg at a private landfill when waste input is 1,360 Mg/day.*

Ultimately, the costs of developing and operating municipal solid waste landfills are covered by user ("tipping") fees, general tax revenues, or a combination of the two.† The use of taxes as a revenue source rather than tipping fees has implications on waste disposal services. First, when disposal costs are included in taxes, most people are not aware of the actual costs involved.[84] Without an effective mechanism for transmitting cost information, waste generators have no incentive to reduce their generation rates. Second, tax-supported facilities are typically underfunded relative to actual disposal costs, resulting in poorer operation than fully funded landfills supported by tipping fees.[85]

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*Differences in the disposal costs at public and private landfills of the same size are attributable to differences in the interest rates available to public and private entities for financing capital expenditures. This is discussed in more detail in Section 8.3.3.

†Initial development costs are usually financed by borrowing money (either through selling bonds or loans). Eventually, the borrowed money is repaid with revenues from tipping fees, general tax revenues, or a combination of the two.

TABLE 8-2. DISPOSAL COSTS PER Mg MSW AT LANDFILLS OF VARIOUS SIZES[83]

MSW accepted		Disposal costs for public owners	Disposal costs for private owners
(Mg/day)	(Mg/yr)	($/Mg)	($/Mg)
10	2,360	71.90	92.20
25	5,900	43.80	58.20
70	17,700	31.00	40.90
160	41,300	16.70	22.20
340	88,400	10.80	14.50
680	177,000	7.97	10.70
1,360	354,000	7.83	10.60

Does not include Subtitle D costs.

Figure 8-11 shows the methods of generating revenues for municipal landfills. Approximately 30% of landfills receive all their revenues from tipping fees, and approximately 35% of landfills receive all their revenues from taxes. The remaining 35% of landfills cover the costs of waste disposal through a combination of tipping fees and taxes.[87]

Factors that influence the choice of revenue sources include landfill size and ownership. Figure 8-12 illustrates the percentage of landfills receiving 80% or more of their revenues from taxes and tipping-fees relative to quantity of waste received. Landfills receiving small quantities of waste are likely to rely heavily on taxes for their revenue while larger landfills rely on both taxes and tipping fees.[89]

Not surprisingly, private owners of landfills rely heavily on tipping-fees relative to other owners of landfills (see Figure 8-13). However, private owners also tend to own larger landfills. It remains unclear whether private landfills rely on tipping fees because they are larger, or larger landfills rely heavily on tipping fees because they are private.[91]

According to the National Solid Waste Management Association, the average tipping fee charged by landfills in 1988 was $29.69 per Mg.[92] This fee is more than twice the average fee charged in 1986.* Although the increase is a reflection of increasing land disposal costs, a distinction must be drawn between tipping fees and the actual costs of landfilling. Communities often set tipping fees to cover current operating costs without regard to amortization of capital expenditures (capital equipment, land, closure, and long-term care costs). Similarly, the cost of disposal for the 35% of landfills supplementing tipping-fee revenues with taxes is usually much higher than the fee charged.[93]

Inefficient landfill pricing may be a major cause of current MSW disposal capacity problems. Dunbar and Berkman[94] and Crew and Kleindorfer[95] claim that tipping fees set below the full marginal cost to society of waste disposal have resulted in waste generation rates greater than if tipping fees equalled marginal cost, because recycling and conservation are rejected in favor of artificially low cost landfilling.

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*Much of the large increase is a result of the addition of sites in the Northeast with high tipping fees.

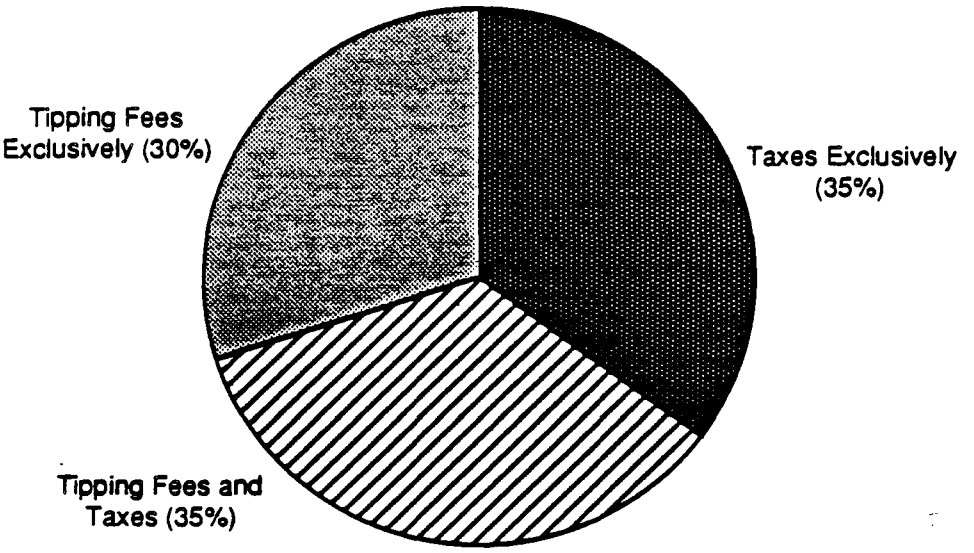

Figure 8-11. Methods of financing municipal solid waste landfilling.[86]

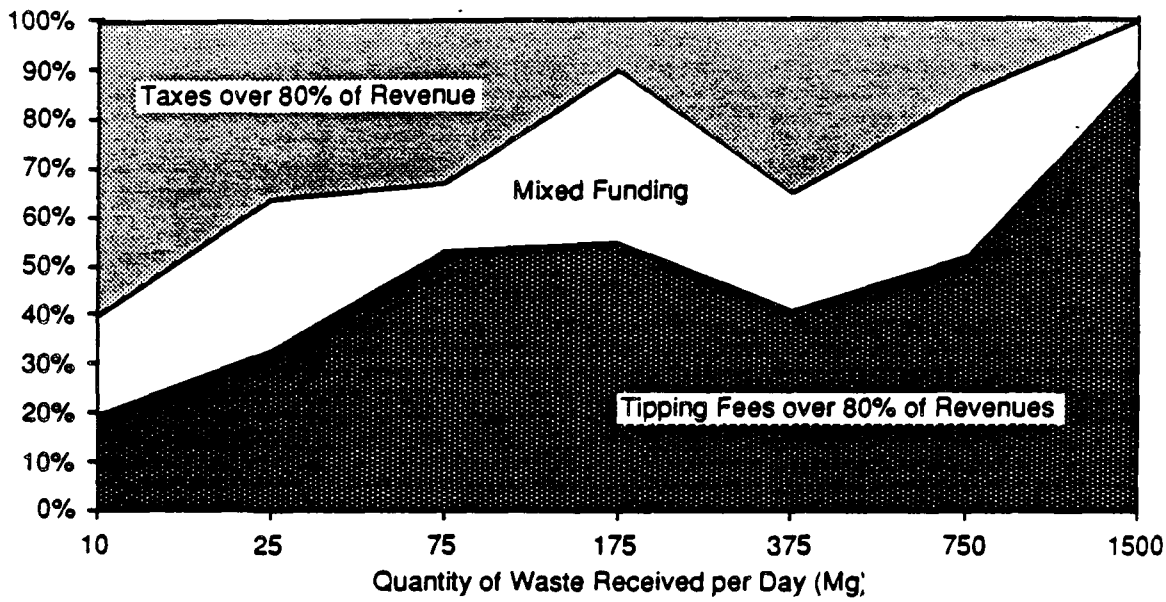

Figure 8-12. Revenue sources for landfill operations by size.[88]

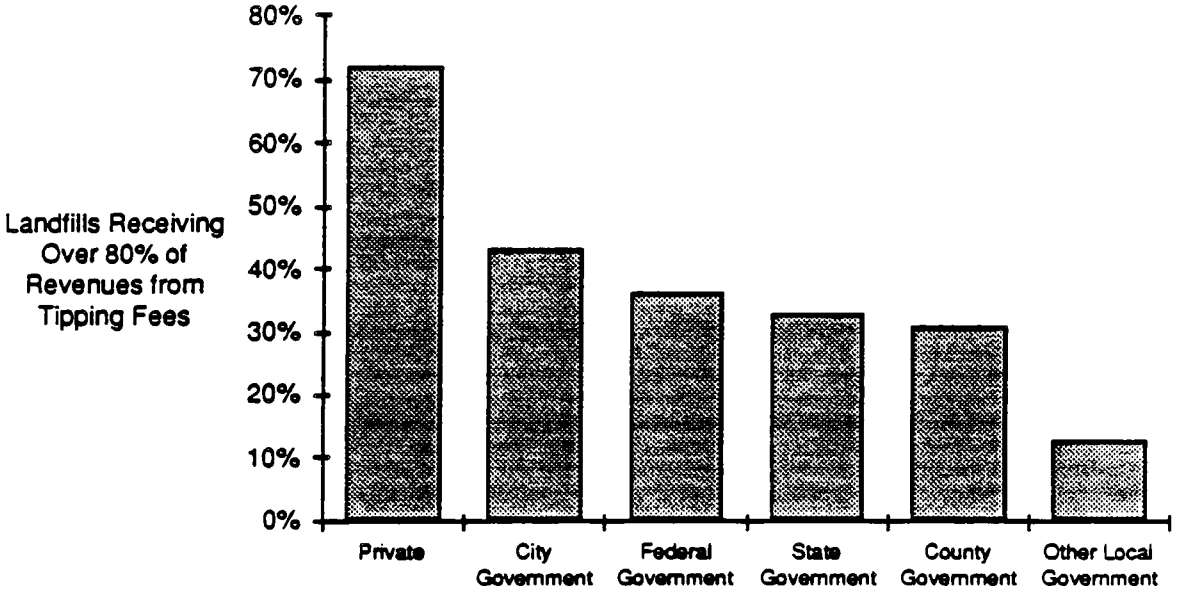

Figure 8-13. Revenue sources for landfill operations by ownership.[90]

Tipping fees lower than marginal social cost have also discouraged private efforts to expand disposal capacity, because of competition from subsidized public landfills.

In addition to tax subsidies, tipping fees do not cover the actual costs to society of disposal because landfill costs usually do no include three important social costs:

- Depletion costs of existing landfills (discounted present value of the difference in landfill costs today and the future costs of a replacement landfill),
- Opportunity costs of land used in landfills, and
- Environmental costs (risk of environmental damage from landfills).

Dunbar and Berkman argue that excluding such costs has contributed to the current crisis in MSW management in major Northeast metropolitan areas.[96]

8.2.4 Changing Regulatory Environment

As the public has become aware of the potential health, safety, and environmental impacts of solid waste management, opposition toward landfilling has increased and a NIMBY attitude toward landfills has become prevalent. Public awareness and concern about the potential impacts of landfilling have placed significant new pressures on state and federal legislators to strengthen regulations on solid waste disposal. As a result, the regulatory environment surrounding landfilling is becoming increasingly stringent, making it a less attractive waste disposal option.

8.2.4.1 Recent and Proposed State Regulations. The pressures of increasing population density, decreasing landfill capacity, and NIMBY attitudes toward MSW management most strongly influence state and local officials. Hence, the changing regulatory environment is most evident at the state level. In 1988 alone, 24 states enacted legislation substantially changing the manner in which MSW is managed.[97]

In recent years, recycling has dominated municipal solid waste legislation. Ten states had mandatory source separation laws by January 1989 with more states expected to follow.[98] One of the most comprehensive source separation program was enacted in New Jersey in 1980. The program's goal was to extend the life of existing landfills by recycling 25% of the municipal solid waste stream by 1986.[99]

Many other states have enacted similar laws encouraging materials recovery. The most common laws establish goal-oriented source reduction programs. Other laws encourage recycling by imposing surcharges on all waste entering landfills or by offering low-interest loans to institutions establishing recycling programs.[100]

States are also moving quickly to establish stringent requirements on landfill construction and operation. Conditions in some states have become regulated to the point that siting new landfills is characterized by some officials as "looking for a needle in a haystack."[101] Examples of state landfill regulatory conditions include:

Connecticut -	Stringency of landfill regulations has prevented new municipal solid waste landfill sitings since 1978.[102]
Florida -	New landfill laws require trained operators, liners, leachate collection systems, groundwater monitoring, and closure plans.[103]
New York -	Landfill design regulations require double composite liners, groundwater monitoring, and leachate collection systems. The rules are considered much more stringent than proposed federal regulations.[104]
Pennsylvania -	New regulations require mini-wastewater treatment plants for leachate management, double liners, and liability insurance. The National Solid Waste Management Association expects the new requirements to force closure of many facilities.[105]
Virginia -	New landfill regulations require double-synthetic liners, groundwater monitoring, and leachate collection.[106]

8.2.4.2 Forthcoming Federal Regulations. Pressures for more stringent landfill regulations have also been felt at the federal level. EPA is currently developing a regulatory program for municipal solid waste landfills under both the Resource Conservation and Recovery Act and the Clean Air Act.

Clean Air Act Regulations and Standards

As explained in this document, EPA's Office of Air Quality Planning and Standards is developing air emissions for closed/existing and new municipal landfills under §111(d) and §111(b) of the Clean Air Act (CAA).

EPA has scheduled proposal of these regulations for 1990. The CAA regulations will limit air emissions of nonmethane organic compounds, air toxics, odors, carbon dioxide, methane, and other explosive gases from landfills. The regulations will require the active collection and disposal of air emissions.

Resource Conservation and Recovery Act Regulations

Subtitle D of the Resource Conservation and Recovery Act (RCRA) regulates municipal solid waste landfills. EPA initially issued criteria for landfills in 1979 that prohibit

- operating a landfill in a floodplain,
- harming endangered species,
- discharging wastewater without permits,
- contaminating groundwater,
- open burning of waste, and
- failing to control disease vectors (i.e., rats).

The 1984 Hazardous and Solid Waste Amendments to RCRA directed EPA to revise these initial landfill criteria to further protect the public health and the environment. The EPA is currently considering new rules regulating the siting, operation, closure, and post-closure of landfill facilities.[107] The rules under consideration restrict the location of new and existing landfills near airports, floodplains, wetlands, fault areas, seismic impact zones, and other unstable areas.

The Subtitle D rules under consideration also impose numerous design and operating criteria on landfills. In many cases, they would significantly change the manner in which landfills operate. They would require:

- daily cover of waste,
- control of disease-vector populations,
- monitoring for explosive gasses in facility structures,
- limiting public access to landfill sites,
- eliminating surface water discharge,
- run-on/run-off water controls,
- extensive record keeping, and
- eliminating leachate recirculation.

Furthermore, the Subtitle D rules under consideration would require a program to detect and prevent the disposal of the following wastes:

- regulated hazardous wastes

- polychlorinated biphenyls (PCBs), and
- bulk and noncontainerized liquids, and containers holding free liquids (unless the liquids are household or septic wastes).

The Subtitle D rules under consideration would also impose extensive new post-closure requirements on landfill owners. Particularly, they would require landfill owners to develop a long-term care plan with a minimum scope of 30 years requiring maintenance and operation of:

- leachate collection systems,
- groundwater monitoring systems,
- final covers, and
- gas monitoring systems.

In addition to more stringent siting, operating, and closure criteria, the rules under consideration would include new groundwater monitoring and corrective action requirements. Furthermore, owners and operators would need to demonstrate the ability to finance closure, long-term care, and any potential corrective action of known contamination.[108]

In conclusion, the new Subtitle D rules under consideration would impose significant new costs on landfill operations. Table 8-3 shows estimates of the costs of the rules on landfills of different sizes. Increases in landfill costs will range from 20 to 40 percent due to the Subtitle D requirements. Not surprisingly, many landfill facilities are expected to close after promulgation of the new rules.

8.2.5 Future Prospects for Municipal Solid Waste Landfilling

Rising costs and increasingly stringent regulations have resulted in many landfill closures. Between 1978 and 1988, an estimated 14,000 landfills, or 70% of landfills operating in 1978 closed. In addition, EPA estimates that one-half of all municipalities will run out of landfill space in 10 years and that one-third will run out within 5 years.[110] Table 8-4 presents the projected closures of existing landfills and the corresponding change in MSW acceptance rate. In 1988, 5,499 landfills handled 187 million Mg of waste. EPA projects that existing landfills still operating in 2013 will only accept 19 million Mg of MSW.[112]

While many landfills have closed in recent years, the number of new facilities opening has experienced a rapid decline. Specifically, the number of facilities opening each year has declined from between 300 and

TABLE 8-3. ESTIMATED RCRA SUBTITLE D COSTS TO LANDFILLS[109]

MSW accepted		Subtitle D criteria costs–public	Cost increase pubilc owners	Subtitle D criteria costs–private	Cost increase private owners
(Mg/day)	(Mg/yr)	($/Mg)	(%)	($/Mg)	(%)
10	2,360	18.50	25.7	23.80	25.8
25	5,900	12.90	23.3	17.70	29.6
70	17,700	7.89	25.5	10.40	25.4
160	41,300	5.98	35.8	7.96	35.9
340	88,400	4.33	40.1	5.84	40.3
680	177,000	2.85	35.8	3.83	35.8
1,360	354,000	2.82	36.0	3.82	36.0

TABLE 8-4. ESTIMATED NUMBER AND ANNUAL ACCEPTANCE RATE OF EXISTING MUNICIPAL LANDFILLS, 1988 TO 2013[111]

Year	Number of landfills	Annual quantity of waste received (10^6 Mg)
1988	5,499	170
1993	3,332	119
1998	2,720	85
2003	1,594	54
2008	1,234	32
2013	1,003	17

400 per year in the early 1970s to between 50 and 200 in the late 1980s, without any accompanying increase in landfill size. If current trends in landfill development continue, only one-third of disappearing capacity will be replaced.[113]

Siting Difficulties

In most states, siting problems have been the major cause of decreasing landfill capacity.[114] Public opposition and a NIMBY attitude are the major obstacles to successful siting of landfills and other waste management facilities. Psychologists suggest that three main factors contribute to the NIMBY syndrome:[115]

- public perceptions of landfills conflict with the "cleanliness ethic" of most individuals,
- landfills may negatively influence the self-image of both the individuals living nearby and their neighborhoods collectively, and
- rural communities that manage their own wastes resent having MSW forced onto them by urban communities who are used to others managing their waste.

Economics also play an important role in landfill siting. A common objection to landfill siting is the impact on the value of nearby properties. Although a 1972 study conducted for the EPA concluded that solid waste disposal sites have no apparent negative effect on property values, other studies have suggested that neighboring properties may experience as much as a 25% reduction in value.[116]

Landfill siting and development problems are most acute in the Northeast. Landfill problems in specific states include:

Connecticut - No new landfills have been sited since 1978,

New Jersey - Landfill shortages have transformed the state into a net exporter of municipal solid waste (MSW exported exceeds MSW disposed inside the state),

Pennsylvania - Cannot find a replacement site for 4,500 Mg/day (approximately 0.6 percent of total national capacity) landfill in Scranton that closed in 1987,[117] and

New York - Unable to site a replacement facility for 24,500 Mg/day (over 3% of total national capacity) landfill in Staten Island that is expected to close within 5 years. Proposals to replace this capacity with combustors are facing public opposition.[118]

However, the landfill capacity problems are not isolated to the Northeast. For example, population growth in Florida indicates a need for an additional 2,700 acres of landfill area annually through 1995. A recent survey of state solid waste management offices by the Association of State and Territorial Solid Waste Management Officials found only four states reporting no capacity problems: Kansas, Nevada, North Dakota, and South Dakota.[119]

Although public opposition in many areas has prevented development of new landfills, several states have experienced success at landfill siting. A survey by the NSWMA identified successful state landfill siting programs in Wisconsin and Delaware.[120]

Wisconsin's landfill siting program is perhaps the most successful. Under the program landfills are being sited at a rate of 10 to 20 per year, making most capacity problems short-term.[121] Wisconsin's program divides the siting process into two steps:

(1) The landfill application is reviewed by the state's Department of Natural Resources for feasibility, necessity, and regulatory compliance, and

(2) After DNR approval and determination of need, other matters (including compensation) are negotiated between the landfill developer and "affected local community".[122]

Wisconsin's siting program is successful for two reasons: (1) the program limits the number of points in the process where a siting can be stopped and the number of reasons why a siting can be stopped; and (2) the program allows for negotiation of almost any characteristic of the landfill short of blocking its development.[123]

Delaware has created the Delaware Solid Waste Authority (DSWA), a government entity charged with siting new landfills. Although the program is criticized for limiting private participation, the program has successfully overcome political pressures and NIMBY attitudes. In the past five years, the DSWA has sited three new landfills (one for each county in the

state) and a combustor. Landfill capacity for each county is now sufficient for an estimated 30 to 40 years.[124]

Decreasing landfill capacity has made importing and exporting trash between states and counties commonplace. For example, over one million tons of MSW is imported and disposed in Ohio landfills. The quantity of MSW exported from New Jersey actually exceeds the quantity of MSW managed inside the state. In the face of landfill siting problems, states and counties with diminishing landfill capacities are taking steps to keep waste generated in other jurisdictions from crossing borders and being disposed in their landfills. At least ten states have enacted legislation limiting or prohibiting waste imports between states or counties:[125]

- Arkansas
- Georgia
- Maryland
- New York
- Pennsylvania
- Delaware
- Kentucky
- New Jersey
- Ohio
- Rhode Island

Various legislation has also been introduced in the U.S. Congress that would ban the transport of waste across state lines or outside the United States. The frequency of waste import bans have led some to observe that a "garbage war exists between the states." Although the constitutionality of waste import bans is questionable, they present serious interim problems for states and counties with diminishing landfill capacity.[126]

Increasing Role of Transportation

Long-distance hauling of MSW has been a primary response to rising landfill disposal costs and increasing public opposition to disposal sites.[127] Examples of municipalities that are required to transport waste very long distances include:[128,129,130]

New York City - MSW is hauled 400 to 500 miles to Ohio and to upstate New York,

Philadelphia - Hauls MSW hundreds of miles to upstate New York and Ohio. Has $9 million contract to ship MWC ash to Panama,

Newark, NJ - Faced with a 400% increase in disposal costs, MSW is hauled to Pennsylvania and upstate New York,

Portland, OR - Preparing to haul MSW by truck, rail, and barge to a landfill 140 miles away in eastern Oregon,

Boston - Hauls MSW over 100 miles to upstate New York.

Hauling MSW long distances can substantially increase MSW disposal costs. For example, Waste Management Inc. estimates that the cost of transporting MSW from New York or Philadelphia to an upstate New York landfill could be as high as $880 ($44/Mg) for a tractor trailer carrying 20 Mg of MSW.[131] This value for transportation costs alone is substantially more than the national average for tipping fees charged at landfills. As less expensive landfills near generators continue to close, expensive long-distance hauls of MSW are likely to become commonplace, resulting in tremendous increases in waste disposal expenditures.

Trend Toward Municipal Waste Combustion

The growing shortage of landfill space and rising landfill costs have forced municipal planners to consider alternative waste management options. Materials recovery is usually viewed as an attractive option because it increases the life of existing landfills by diverting potentially large quantities of spent materials into reuse.[132] Materials recovery also slows the depletion of natural resources and removes toxic materials from the waste stream prior to disposal, enhancing the safety of landfilling and combustion.[133]

However, most attention toward alternatives to landfilling has been given to municipal waste combustion (MWC). As noted earlier, municipal waste combustion is the process of reducing the volume of MSW through incineration. Combustion of MSW is attractive because it reduces waste volume by as much as 90%. Therefore, many municipal planners view MWC as an important means of extending the lives of existing landfills and reducing the need for additional landfill capacity.

Table 8-5 presents the historical and projected shares of MSW managed in MWCs from 1960 to 2000. In 1960 the entire municipal solid waste stream was either landfilled or recycled; no waste was incinerated. By 1986, the quantity of MSW incinerated in municipal waste combustors had risen to six percent of the total waste stream. Franklin Associates projects that by the year 2000 about 17% of the MSW stream will be incinerated.

However, municipal waste combustion has many problems similar to landfilling. MWC siting problems have been significant and have prevented

TABLE 8-5. HISTORICAL AND PROJECTED SHARES OF MUNICIPAL SOLID WASTE MANAGED IN MUNICIPAL WASTE COMBUSTORS[134]

Year	Gross discards of MSW (10^6Mg)	Combustion of MSW (10^6Mg)	Share of gross discards (%)
1960	79.4	0.0	0.0
1965	92.8	0.2	0.2
1970	109.3	0.4	0.4
1975	113.6	0.6	0.5
1980	129.4	2.4	1.9
1981	131.3	2.1	1.6
1982	128.8	3.2	2.5
1983	134.6	4.5	3.3
1984	139.3	5.9	4.2
1985	138.3	6.9	5.0
1986	143.0	8.7	6.1
1990[a]	151.8	12.1	8.0
1995[a]	163.4	20.4	12.5
2000[a]	174.8	29.0	16.6

[a]Projection.

siting in several locations.[135] Additionally, the EPA is considering new air emission regulations that would significantly increase the capital and operating costs of municipal waste combustion.[136] Similarly, legislation has been introduced in the U.S. Congress that would require ash from MWCs to be treated as hazardous waste and disposed in hazardous waste landfills.[137] According to a Kidder Peabody report, more MWC capacity was canceled than was ordered in 1987, resulting in a 10% decline in projected capacity.[138]

8.3 REGULATORY ALTERNATIVES AND CONTROL OPTIONS

8.3.1 Regulatory Alternatives

As explained in detail in Chapter 5, EPA is considering regulatory alternatives for controlling air emissions from two types of municipal solid waste landfills: closed/existing landfills and new landfills. EPA will control emissions from closed/existing landfills under the guidelines of §111(d) of the Clean Air Act (CAA) and will regulate emissions from new landfills under CAA §111(b).

EPA will require closed/existing and new landfills to install and operate emissions controls as long as their annual nonmethane organic compound (NMOC) emission rates exceed a specified cutoff level. In other words, landfills must install emission controls once NMOC emissions reach a specified cutoff level, and they must continue controlling NMOC emissions until they drop below the specified cutoff, which may be many years after closure. EPA is evaluating three possible cutoff levels for NMOC emissions from closed/existing and new landfills: 25, 100, and 250 Mg of NMOC per year. The 25 Mg NMOC/yr cutoff level is the most stringent, while the 250 Mg NMOC/yr is the least stringent, because the former requires emissions controls for lower levels of emissions than the latter.

8.3.2 Emissions Control Options

Chapter 4 describes the two basic emissions control approaches for landfills: combustion without energy recovery (i.e., flares) and energy recovery (mainly involving the combustion of the landfill gas to produce steam or electricity). For simplicity, we refer to these two control approaches as the flare option and the energy recovery option. So landfills exceeding specified NMOC emission rates will have a choice

between the flare option or the energy recovery option for controlling emissions. The remainder of this chapter emphasizes the economic impacts of the various NMOC stringency levels assuming that all affected landfills choose the flare option. However, we also discuss some of the economic impacts of the energy recovery option. Appendix F contains the tables evaluating the energy recovery option.

The assumption that all affected landfills choose the flare option results in overestimates of the actual costs of the regulatory alternatives for two reasons. First, the affected landfills that would have installed energy recovery equipment in the absence of EPA emissions regulations should be excluded from cost estimates for such regulations. Second, it will be cheaper for some of the other affected landfills to install energy recovery equipment rather than flares, because the revenues from energy recovery will exceed the extra cost of the energy recovery equipment. So the costs of the flare option will overestimate the costs actually incurred at these landfills. These two reasons are discussed in more detail below.

As indicated in Section 8.2, some landfills in recent years have installed energy recovery equipment in the absence of EPA emissions regulations, because they expect the revenues from the sale of electricity (or steam or medium/high Btu gas) to exceed the cost of the energy recovery. In other words, these landfills expect their energy recovery efforts to make a profit. Presumably, some landfills in the future would have also installed energy recovery equipment in the absence of EPA regulations. Theoretically, these landfills would be excluded from the group of landfills affected by EPA's emissions control regulations, because they would be controlling their emissions in the absence of such regulations. So neither the flare costs nor the NMOC emission reductions at these landfills should be attributed to the EPA emissions regulations.

There is no way to precisely determine which landfills would have installed energy recovery equipment in the absence of EPA emissions regulations. First, the acceptance of new technologies (such as energy recovery from landfill gas) often spreads slowly. Consequently, some landfills that would profit from energy recovery may not choose this option as a result of a general aversion to new technologies. Second, energy

recovery requires more capital equipment than flaring landfill gases. Since revenues from energy recovery involve some uncertainty, some landfill owners may have difficulty getting the additional capital required for energy recovery, or they may not want to risk the additional capital on this option. Finally, landfill gas generation rates depend on factors such as the amount and composition of MSW going into landfills over time and rainfall. The model in Chapter 7 assumes values for many of these factors. To the extent that the actual values for these factors differ from the assumed values, the model may overestimate or underestimate the profitability of energy recovery at particular landfills. In these cases, the model's predictions of which landfills will choose an energy recovery option in the absence of EPA emissions regulations may not be accurate.

An EPA emissions regulation will probably stimulate the adoption of energy recovery at some landfills, because such a regulation will lower the cost of this option. In particular, evaluating the feasibility of energy recovery requires costly information on the characteristics and flow of landfill gases. The EPA regulatory alternatives under consideration will require many landfills to test their landfill gases in order to determine the need for controlling NMOC emissions. Thus, many landfills will have to collect the landfill gas information that is needed to evaluate the energy recovery option. Furthermore, the flare and energy recovery options can use the same wells and collection system. Therefore, landfills that must control their NMOC emissions will need to install wells and a collection system that they could also use for energy recovery. Having already incurred the costs of getting information on landfills gases and installing wells and a collection system, the additional costs of the energy recovery option are relatively small. Thus, some (possibly many) landfills will choose an energy recovery option that would not have chosen this option in the absence of the EPA regulation.

For our analysis of the energy recovery option, we assume that landfills showing a profit from energy recovery (based on the model described in Chapter 7) would have installed energy recovery equipment in the absence of EPA emissions regulations. Furthermore, we also assume that the landfills that will not make a profit from energy recovery will select

the least-cost emissions control option. In some cases the flare option will have the lowest cost, while in other cases the revenues from energy recovery will result in the energy recovery option having a lower net cost than the flare option. In summary, our energy recovery option reports the cost-minimizing control option (either flares or energy recovery) only for landfills with energy recovery costs that exceed energy recovery revenues (i.e., landfills with positive energy recovery costs).

In conclusion, the flare option overestimates the actual cost of the regulatory alternatives, because some landfills will install cheaper emissions controls voluntarily. On the other hand, the energy recovery option will underestimate compliance costs when:

- landfills that would have implemented an energy recovery option in the absence of EPA emissions regulations are required to control their emissions longer than they would voluntarily (i.e., when the required emissions control period is longer than the profit maximizing energy recovery period), and

- landfills that the model predicts would profit from energy recovery decide to install flares (for reasons discussed above) in order to comply with the EPA emissions regulation.

However, the energy recovery option will overestimate compliance costs at landfills that select an energy recovery option as a result of the EPA emissions regulation and make a profit, but would not have installed energy recovery in the absence of EPA emissions regulations (because they did not realize that they would profit from energy recovery, for example). The aggregate result of these opposing tendencies is unknown.

8.3.3 Additional Assumptions and Their Implications

The model described in Chapter 7 has two features that lead to overestimates of the number of landfills affected by the §111(d) and 111(b) regulatory alternatives under consideration and the compliance costs and emissions reductions at the affected landfills for each of the alternatives. These features are:

- the model assumes that every landfill that closes after 1986 is replaced by a landfill having identical characteristics, and

- the model uses data on individual landfills that overestimate the total amount of MSW going to landfills each year.

We discuss each of these features below.

As indicated in Section 8.2, over half of the 6,034 active landfills in 1986 are expected to close by the year 1998. At the same time very few new landfills are being developed (for reasons discussed previously). Consequently, the total number of landfills in the United States is declining. On the average, new landfills are not larger than the closing landfills, so landfill capacity is also declining as the number of landfills falls.

At the current time there is no method for predicting which landfills that close will be replaced by new landfills. Furthermore, there is no method for determining the characteristics of the replacement landfills (such as their design capacity). Since the number and characteristics of replacement (i.e., new) landfills are needed for the costing model in Chapter 7, it is assumed that every landfill that closes between 1987 and 1997 is replaced by a landfill having identical characteristics to the landfill that closed.* Since there will probably be fewer landfills, this assumption tends to overestimate the number of landfills affected by the regulatory alternatives under consideration. This leads to overestimating the total cost of the regulatory alternatives.

In 1986 EPA's Office of Solid Waste (OSW) conducted a survey of municipal landfills in the United States (as discussed above). This survey obtained extensive information on the characteristics of landfills, such as their design capacity, year of opening, anticipated year of closing, refuse in place at the end of 1986, and the amount of MSW received in 1986. The cost model in Chapter 7 uses data from the OSW landfills survey in determining which landfills are affected by the regulatory alternatives under consideration and the compliance costs and emissions reductions for each affected landfill for these alternatives. In particular, the amount of MSW received in 1986 is an important variable in determining compliance costs and emissions reductions.

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*Landfills that close between 1987 and 1992 are categorized as "closed" landfills. "Existing" landfills include landfills that will close sometime after 1992 and landfills that replace landfills that close between 1987 and 1992. "New" landfills are landfills that replace landfills that close between 1992 and 1997. See Chapter 3 for additional details on these designations.

The OSW landfills data were collected to evaluate regulatory alternatives under Subtitle D of RCRA. There were two difficulties in using the OSW landfills data to analyze control costs and emissions reductions for CAA §111(d) and 111(b) regulatory alternatives. One difficulty was the conversion of cubic yards of MSW into tons. The other difficulty involved differences in historical MSW acceptance rates and the 1986 acceptance rate. The resolution of these difficulties resulted in MSW acceptance rates that are useful on an individual landfill basis but substantially overestimate national MSW generation.

In conclusion, the two factors just discussed lead to overestimates of the number of landfills affected by the §111(d) and 111(b) regulatory alternatives under consideration and overestimates of the national costs and emissions reductions of controls at these affected landfills. So the actual economic impacts of these regulatory alternatives will probably be less than the economic impacts described in the remainder of this chapter.

Two other assumptions underlying the economic analysis of the regulatory alternatives under consideration are noteworthy. First, we assume that different discount rates are appropriate for publicly owned landfills and privately owned landfills when evaluating the costs of emissions controls from the perspective of landfill owners, which we designate as enterprise costs. As explained in detail in Appendix A of Morris et al., expenditures by public entities have a lower opportunity cost than expenditures by private entities.[139] In particular, we use a 4% discount rate for the capital and operating costs of compliance for publicly owned landfills and a 8% discount rate for compliance costs at privately owned landfills. This results in publicly owned landfills having a higher net present value of enterprise costs than privately owned landfills for the same stream of compliance costs over the same time period.

Following recent EPA guidelines, we use a 2-stage discounting approach for calculating compliance costs from a social perspective. Under this approach capital costs are annualized over the years that controls are operated (i.e., the control period) using a 10% rate for all landfills (regardless of ownership). Then the resulting annualized capital costs and the annual operating costs for all landfills are discounted using a 3%

rate. Kolb and Scheraga explain the rationale for calculating the social costs of compliance using this 2-stage discounting approach.[140]

A second important assumption in the economic analysis is that publicly owned landfills have more flexibility in generating the revenues to pay for the capital and operating costs of emissions controls than privately owned landfills. Specifically, public entities can generate the revenues for compliance costs by increasing taxes of various types or by increasing user fees at the landfill while it is still accepting MSW. Alternatively, private landfills can only cover compliance costs by increas-ing user fees during the landfill's operating life.*

The difference in public and private landfills regarding their ability to generate the revenues for covering the costs of emissions controls has important implications for the annualization period for such costs. In particular, we annualize the enterprise costs for publicly owned landfills over the control period. Even though the landfill will be closed during some of the control period, the public entity that owns the landfill will still be able to tax former users of the landfill (and possibly others) in order to cover the compliance costs. Alternatively, we annualize enterprise costs for privately owned existing landfills over the period from 1992 (the anticipated promulgation date of the regulatory alternative selected) to the landfill's closure date.† We assume that these landfills must sufficiently increase user fees during this time period to cover compliance costs over the entire control period (including the years after closure). Thus, the necessary increase in user fees may be quite large whenever compliance costs are relatively high and the number of years until closure is relatively small.

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*The difference in the ability of public versus private landfills to generate revenues for compliance costs is particularly significant for affected landfills that are closed before the regulations are promulgated. Public entities that own a closed landfill can increase taxes on households and businesses that were previously served by the closed landfill in order to pay for emissions controls. Owners of private landfills that are closed have no way to generate revenues to cover the costs of emissions controls.

†We annualize enterprise costs for privately owned new landfills over the entire operating life of these landfills.

8.4 ANALYSIS OF ECONOMIC IMPACTS

As described in Section 8.3, the EPA is considering regulatory alternatives for controlling air emissions from both closed/existing landfills and new landfills. Section 8.4 first discusses the economic impacts of the three possible nonmethane organic compound (NMOC) emissions level cutoffs under the Guidelines of §111(d) of the Clean Air Act (CAA). Then, this section discusses the economic impacts of proposed regulations under the Standards of CAA §111(b) for the same three possible NMOC emissions level cutoffs. In evaluating the impacts of controls under each section of the CAA, we consider two basic control options: combustion without energy recovery (the flare option), and energy recovery (the energy recovery option).

As described above, increasing NMOC emissions cutoffs (i.e., 25 Mg NMOC/yr, 100 Mg NMOC/yr, and 250 Mg NMOC/yr) represent decreasing levels of stringency for the controls. Thus, for example, more landfills are affected by each control option at the 25 Mg level than at the 100 Mg level. Landfills will be required to operate controls in every year for which their emissions level exceeds the chosen cutoff level. So some landfills may need to operate controls for many years after closure, until the NMOC emissions fall below the chosen cutoff level.

8.4.1 Section 111(d) Guidelines

Guidelines under §111(d) of the CAA address existing sources of emissions. In the case of landfills, these guidelines will apply to both closed and existing landfills, since the level of NMOC emissions builds throughout the active life of a landfill and continues after closure. As indicated in Section 8.3, the model used to estimate emissions assumes that each landfill that closes is replaced by another identical landfill serving the same area.

We first characterize the landfills affected under each stringency level for the flare option, then we address the economic impacts of the stringency levels on affected landfills. Next, we examine the energy recovery option, characterizing the affected landfills under each stringency level and estimating the economic impacts of that option.

8.4.1.1 Flare Option. Under the flare option, landfills are assumed to control their NMOC emissions by collecting the NMOCs and then burning

them, with no provision for energy recovery. We assume that all landfills generating NMOC emissions above a given stringency level are affected by the §111(d) Guideline. As mentioned above, three possible stringency levels are being evaluated: 25 Mg NMOC/yr, 100 Mg NMOC/yr, and 250 Mg NMOC/yr.

Of 7124 landfills (6034 existing landfills and 1090 closed landfills) eligible for coverage under the §111(d) Guidelines, between 5% and 26% of the landfills would be affected, depending on the stringency level selected. As indicated in Table 8-6, if the most stringent 25 Mg NMOC/yr cutoff were selected, 1884 landfills would be affected. If the 100 Mg NMOC/yr stringency level were selected, 853 landfills would be affected, while only 386 landfills would be affected if the 250 Mg NMOC/yr stringency level were selected.

In addition to the total number of affected landfills, Table 8-6 shows a distribution of affected landfills by design capacity under each of the possible stringency levels. Under the most stringent 25 Mg stringency level, a larger proportion of the total number of affected landfills is small (27% have less than 1 million Mg design capacity, 71% have less than 5 million Mg design capacity) than under the less stringent cutoff levels. Only 16% of the affected landfills would have a design capacity below 1 million Mg under the 100 Mg stringency level, while only 6% would fall into this smallest size category under the least stringent 250 Mg cutoff level.

As mentioned above, some landfills will be required to operate emissions controls for many years after they close. This is of particular concern for private landfills, since increased user fees while they are still active and accepting MSW are their only means of paying for these controls. The bottom part of Table 8-6 shows the number of affected privately owned landfills under each stringency level. The landfills expected to have the greatest difficulty paying for the NMOC controls are those which are privately owned and already closed. For these landfills, there exists no possibility of recovering the costs of compliance through increased user fees. As shown by the last line, 4% of the affected landfills under the most stringent 25 Mg level are privately owned closed landfills. Under the 100 Mg stringency level, 6% of the affected landfills

TABLE 8-6. SUMMARY INFORMATION FOR AFFECTED CLOSED AND EXISTING LANDFILLS

	Stringency Levels (Mg NMOC/yr)		
	25	100	250
Number of affected landfills	**1,884**	**853**	**386**
(Percent of total closed and existing landfills)	(26)	(12)	(5)
Distribution of affected landfills by design capacity (10^6 Mg)			
≤ 1	**514**	**133**	**22**
	(27)	(16)	(6)
1 to 5	**837**	**349**	**181**
	(44)	(41)	(47)
5 to 10	**295**	**176**	**48**
	(16)	(21)	(12)
> 10	**238**	**195**	**135**
	(13)	(23)	(35)
Total	**1,884**	**853**	**386**
	(100)	(100)	(100)
Privately owned affected landfills	**406**	**210**	**121**
(Percent of affected landfills)	(22)	(25)	(31)
Existing	**334**	**162**	**82**
Closed	**72**	**48**	**39**

Note: The numbers in parentheses are percentages. Details may not add to totals due to rounding.

are privately owned closed landfills, while under the 250 Mg level, 10% are privately owned and closed.

As noted earlier, landfills will be required to operate emissions controls as long as their NMOC emissions exceed the selected cutoff level. In general, different landfills will reach a given emissions cutoff level in different years. Similarly, the number of years that emissions will exceed the cutoff level will vary from landfill to landfill, and therefore the year that controls may be removed will vary from landfill to landfill. Thus, the possible economic impacts of the emissions controls will be incurred by various landfills during different time periods.

Table 8-7 depicts the distribution of the length of the control period for affected closed and existing landfills under each of the three stringency levels. In general, the control periods range from one to more than 277 years, with the maximum length of control period being slightly longer as the stringency of control increases. The average length of control period ranges from 66 years for the 100 Mg stringency level to 79 years for the 25 Mg stringency level.

As mentioned above, the ease with which landfills will be able to recapture the costs of installing and operating the controls will decrease after the landfill closes. Until that time, the landfill may increase its user fees to offset some of its increased costs. After closure, the public owners of the landfill will have to find some other means of raising revenues (such as taxes), while the private owners will not be able to raise revenues at all. Private landfills must therefore increase user fees sufficiently to offset all their control costs while the landfill is still accepting MSW. Thus, the shorter the length of time between the start of controls and landfill closure, the greater the financial burden of a given control cost on a landfill, especially if it is privately owned.

Table 8-8 provides information about the length of control period prior to closure for all affected closed and existing landfills, and 8-9 provides such information for privately owned affected landfills. The 22% to 23% of affected landfills that are privately owned under the 25 Mg and 100 Mg stringency levels, respectively, have slightly longer control periods prior to closure than the publicly owned affected landfills, while the 27% of affected landfills which are privately owned under the 250 Mg

TABLE 8-7. LENGTH OF CONTROL PERIOD FOR AFFECTED CLOSED AND EXISTING LANDFILLS

	Stringency Levels (Mg NMOC/yr)		
	25	100	250
Average length of control period (years)	79.2	66.3	67.8
Distribution of affected landfills by length of control period (years)			
≤ 25	298 (16)	244 (29)	94 (24)
26 to 50	305 (16)	165 (19)	46 (12)
51 to 100	607 (32)	229 (27)	150 (39)
101 to 150	582 (31)	157 (18)	80 (21)
> 150	92 (5)	58 (7)	16 (4)
Total	1,884 (100)	853 (100)	386 (100)

Note: Numbers in parentheses are percentages. Details may not add to totals due to rounding.

TABLE 8-8. LENGTH OF CONTROL PERIOD PRIOR TO CLOSURE FOR AFFECTED EXISTING LANDFILLS

	Stringency Levels (Mg NMOC/yr)		
	25	100	250
Average length of control period prior to closure (years)	20.4	17.7	19.7
Distribution of affected landfills by length of control period prior to closure (years)			
≤ 5	370 (24)	239 (34)	94 (31)
6 to 10	244 (16)	99 (14)	29 (10)
11 to 20	513 (34)	227 (33)	106 (35)
21 to 50	261 (17)	63 (9)	41 (13)
> 50	133 (9)	70 (10)	32 (11)
Total	1,521 (100)	698 (100)	302 (100)

Note: Numbers in parentheses are percentages. Details may not add to totals due to rounding. Excludes closed landfills.

TABLE 8-9. LENGTH OF CONTROL PERIOD PRIOR TO CLOSURE FOR AFFECTED EXISTING LANDFILLS: PRIVATE LANDFILLS ONLY

	Stringency Levels (Mg NMOC/yr)		
	25	100	250
Average length of control period prior to closure (years)	23.0	20.1	17.0
Distribution of affected landfills by length of control period prior to closure (years)			
≤ 5	73 (22)	56 (35)	22 (27)
6 to 10	39 (11)	17 (10)	10 (12)
11 to 20	130 (39)	53 (33)	29 (35)
21 to 50	46 (14)	19 (12)	19 (23)
> 50	.46 (14)	17 (10)	2 (3)
Total	334 (100)	162 (100)	82 (100)

Note: Numbers in parentheses are percentages. Details may not add to totals due to rounding. Excludes closed landfills.

stringency level have a slightly shorter control period prior to closure. Of particular concern may be the privately owned landfills with ten years or less between the imposition of controls and closure. These comprise 112 of the privately owned affected landfills under the 25 Mg stringency level, 73 under the 100 Mg stringency level, and 32 under the 250 Mg stringency level.

One measure of the cost of complying with the regulatory alternatives under consideration is the net present value of enterprise costs. This measure, shown in Table 8-10, is computed by discounting the flow of capital and operating costs to arrive at a measure of the current value of the costs that will be incurred throughout the control periods for the various landfills. Since most landfills will begin and end controls at different times, using a net present value measure of costs is the appropriate way to compare costs between landfills.

As explained in Section 8.3, the interest rates faced by public owners of landfills differ from those faced by private owners, so we discount the stream of capital and operating costs using a different discount rate for each ownership group. We discount the capital and operating costs incurred by public landfill owners as a result of complying with the regulatory alternatives under consideration using a 4% discount rate, while we discount costs incurred by private landfill owners to their present value using an 8% discount rate. Table 8-10 presents these costs, along with a distribution of the number of affected landfills in several enterprise cost categories for each of the three stringency levels.

The maximum net present value (NPV) of enterprise costs incurred by any landfill is $61 million under the 25 Mg stringency level, $54 million under the 100 Mg stringency level, and $51 million under the 250 Mg stringency level. When summed across all landfills affected by controls under each stringency level, the national total NPV of enterprise costs ranges from $1.93 billion under the 250 Mg stringency level to $5.86 billion under the 25 Mg stringency level (see Table 8-10). A larger proportion of affected landfills incurs a relatively low NPV of enterprise costs ($3 million or less) under the 25 Mg level than under the 100 Mg level or the 250 Mg level. The mean NPV of enterprise costs per affected landfill under the 250 Mg stringency level, $5.00 million, exceeds that for the other two stringency levels.

TABLE 8-10. NET PRESENT VALUE OF ENTERPRISE COSTS FOR AFFECTED CLOSED AND EXISTING LANDFILLS

Net Present Value	Stringency Levels (Mg NMOC/yr) 25	100	250
National enterprise costs (10^6)			
Capital	2,233	1,618	871
Operating	3,625	2,015	1,058
Total	5,858	3,634	1,929
Average total enterprise cost per affected landfill (10^6)	3.11	4.26	5.00
Distribution of affected landfills by net present value of enterprise costs (10^6)			
≤ 0.5	119	60	15
	(6)	(7)	(4)
0.5 to 1.0	165	90	19
	(9)	(11)	(5)
1.0 to 3.0	1,060	341	169
	(56)	(40)	(44)
3.0 to 5.0	331	205	101
	(18)	(24)	(26)
5.0 to 10.0	161	111	53
	(8)	(13)	(14)
>10.0	48	46	29
	(3)	(5)	(7)
Total	1,884	853	386
	(100)	(100)	(100)

Note: Numbers in parentheses are percentages. Net present value of enterprise cost is calculated using a 4 percent discount rate for publicly owned landfills and an 8 percent discount rate for privately owned landfills. Details may not add to totals due to rounding.

Table 8-11 shows another measure of enterprise costs. The annualized enterprise control cost per Mg of MSW for affected existing landfills is computed based on each landfill's NPV of enterprise costs. These costs are annualized using the following formula:

$$\frac{\text{NPV enterprise costs}}{(1 - (1+r)^{-t})/r}$$

where r is the interest rate and t is time.

The interest rate and the length of time over which costs are annualized depend on the ownership of the landfill. As explained previously, publicly owned landfills are annualized using a 4% interest rate over the time period during which controls will be in place. Privately owned landfills, on the other hand, will not be able to recapture their compliance costs after they stop accepting MSW. The enterprise costs for privately owned landfills, therefore, are annualized over the period from 1992 until the landfill closes, using an 8% interest rate.

To compute the annualized enterprise cost per Mg of MSW for affected existing landfills, the annualized cost is divided by the quantity of waste accepted by the landfill in 1986.* One measure of the average annualized cost per Mg of waste accepted is the national annualized cost per Mg of MSW, which is computed for each stringency level by summing the annualized enterprise costs for all the affected landfills at that level, and then dividing by the summed quantities of waste accepted by all the affected landfills in 1986. The national average annualized costs per Mg of MSW at each stringency level is less than $1 per Mg. These national annualized costs per Mg of MSW range from $0.72/Mg at the 250 Mg stringency level to $0.89/Mg at the 25 Mg level.

Table 8-11 also contains a frequency distribution of affected landfills by annualized cost per Mg of MSW accepted in 1986. The frequency distribution indicates that the proportion of affected landfills incurring annualized costs of $1.25 per Mg of MSW or less increases as the level of stringency decreases. At the 25 Mg stringency level, about 45% of

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*As noted in Section 8.3, the historical annual average amount of MSW accepted by the landfill is substituted for the quantity of MSW received in 1986 for some landfills.

TABLE 8-11. ANNUALIZED ENTERPRISE CONTROL COST PER Mg OF MSW FOR AFFECTED EXISTING LANDFILLS

	Stringency Level (Mg NMOC/yr)		
	25	100	250
National annualized cost per Mg MSW ($/Mg MSW)	0.89	0.84	0.72
Distribution of affected landfills by annualized cost per Mg MSW ($/Mg MSW)			
≤ 0.50	207 (14)	135 (19)	77 (25)
0.50 to 1.25	474 (31)	220 (32)	106 (35)
1.25 to 3.00	426 (28)	206 (30)	92 (30)
3.00 to 10.00	320 (21)	123 (18)	27 (9)
> 10.00	94 (6)	14 (2)	0 (0)
Total	1,521 (100)	698 (100)	302 (100)

Note: Numbers in parentheses are percentages. Costs for publicly owned landfills are annualized at 4 percent over the control period. Costs for privately owned landfills are annualized at 8 percent from 1992 to the year of closure. Details may not add to totals due to rounding. Excludes closed landfills.

landfills experience annualized costs of $1.25 per Mg or less; the maximum annualized cost at this level of stringency, however, is $57 per Mg. At the 100 Mg stringency level the maximum annualized cost falls to $25 per Mg of MSW, and the proportion experiencing costs of $1.25 per Mg or less increases to 51%. Finally, at the 250 Mg stringency level, 60% of affected landfills experience annualized costs per Mg of MSW of $1.25 or less, and the maximum annualized cost experienced is only $8 per Mg.

As noted above, the enterprise costs for privately owned landfills are annualized over a period beginning when the regulation takes effect in 1992 and ending when the landfill closes. Privately owned landfills can only recapture their costs through increased user fees while they are still accepting MSW. The shorter the period of time between 1992 and the year the landfill closes, therefore, the greater the potential burden of a particular amount of control costs on the landfill's owners. Tables 8-12 and 8-13 give the same information as Table 8-11, but for privately owned landfills which have five or fewer years until closure or 5 to 10 years until closure, respectively. Table 8-12 shows that the national annualized enterprise cost per Mg of MSW accepted for private landfills with five years or less until closure is more than five times the national annualized costs for all affected landfills at each stringency level. Specifically, at the 250 Mg stringency level, the national annualized enterprise cost is $5.33 per Mg of MSW, it is $4.37 per Mg of MSW at the 100 Mg level, and it is $5.24 per Mg at the 25 Mg stringency level. At the 100 Mg stringency level, 90% of the 41 affected landfills that are expected to close by 1997 experience annualized costs between $3.00 and $10.00 per Mg of MSW.

For private landfills closing between 1998 and 2002, unit control costs are not nearly as high as the unit control costs of private landfills closing before 1988 (see Table 8-13). The national average measure is $1.17/Mg of MSW at the 25 Mg stringency level, $0.95/Mg of MSW at the 100 Mg stringency level, and only $0.48/Mg at the 250 Mg stringency level. At the 250 Mg stringency level, only two landfills affected are expected to close between 5 and 10 years after 1992, and they incur costs less than $0.50 per Mg of MSW. At the 100 Mg stringency level, only 7 affected landfills are expected to close between 1998 and 2002, and they experience annualized enterprise costs between $0.50/Mg and $1.25/Mg. At the 25 Mg

TABLE 8-12. ANNUALIZED ENTERPRISE CONTROL COST PER Mg OF MSW FOR AFFECTED EXISTING LANDFILLS WITH DATE OF CLOSURE BEFORE 1998: PRIVATE LANDFILLS ONLY

	Stringency Level (Mg NMOC/yr)		
	25	100	250
National annualized cost per Mg MSW ($/Mg MSW)	5.24	4.37	5.33
Distribution of affected landfills by annualized cost per Mg MSW ($/Mg MSW)			
≤ 0.50	0 (0)	0 (0)	0 (0)
0.50 to 1.25	2 (4)	2 (5)	0 (0)
1.25 to 3.00	2 (4)	2 (5)	2 (14)
3.00 to 10.00	39 (67)	37 (90)	12 (86)
> 10.00	15 (26)	0 (0)	0 (0)
Total	58 (100)	41 (100)	14 (100)

Note: Numbers in parentheses are percentages. Costs for privately owned landfills are annualized at 8 percent from 1992 to the year of closure. Details may not add to totals due to rounding. Excludes closed landfills.

TABLE 8-13. ANNUALIZED ENTERPRISE CONTROL COST PER Mg OF MSW FOR AFFECTED EXISTING LANDFILLS DATE OF CLOSURE BETWEEN 1998 AND 2002: PRIVATE LANDFILLS ONLY

	Stringency Level (Mg NMOC/yr)		
	25	100	250
National annualized cost per Mg MSW ($/Mg MSW)	**1.17**	**0.95**	**0.48**
Distribution of affected landfills by annualized cost per Mg MSW ($/Mg MSW)			
≤ 0.50	**0** (0)	**0** (0)	**2** (100)
0.50 to 1.25	**10** (59)	**7** (100)	**0** (0)
1.25 to 3.00	**0** (0)	**0** 0	**0** (0)
3.00 to 10.00	**7** (41)	**0** (0)	**0** (0)
> 10.00	**0** (0)	**0** (0)	**0** (0)
Total	**17** (100)	**7** (100)	**2** (100)

Note: Numbers in parentheses are percentages. Costs for privately owned landfills are annualized at 8 percent from 1992 to the year of closure. Details may not add to totals due to rounding. Excludes closed landfills.

level, 17 landfills are expected to close between 1998 and 2002, with annualized costs between $0.50/Mg and $10.00/Mg.

Table 8-14 presents the annualized enterprise cost per household for affected existing landfills. This attempts to assess the annualized cost that will be borne by households served by affected landfills. To compute this measure, the annualized enterprise costs are divided by an estimated number of households served by the affected landfills.* The national annualized enterprise cost per household for each stringency level is computed by summing the annualized enterprise costs incurred by all affected landfills at that stringency level, and then dividing by an estimate of the total number of households served by those landfills in 1986. The national annualized enterprise cost ranges from $4.16 per household at the 250 Mg stringency level to $5.18 per household at the 25 Mg stringency level. At the intermediate 100 Mg stringency level, the national annualized enterprise cost is $4.90 per household.

The frequency distribution of affected landfills by annualized enterprise cost per household, also shown in Table 8-14, indicates that one-fifth of affected landfills at the 25 Mg stringency level will incur annualized enterprise costs of $3.50 per household or less, and 43% will incur annualized enterprise costs of $7.00 per household or less, although the maximum annualized cost at this stringency level is $332 per household. At the 100 Mg stringency level, the maximum annualized cost incurred is $148 per household; however, one-quarter of the affected landfills will incur annualized costs of $3.50 per household or less and one-half will incur costs of $7.00 per household or less. Only 10% of affected landfills will incur annualized costs of $30.00 per household or more under the 100 Mg stringency level. At the 250 Mg stringency level, over one-third of

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*We estimated the number of households served by affected landfills using the amount of MSW received by these landfills and an average amount of MSW generated by households. We calculated the latter by dividing the total amount of MSW going to all landfills based on the OSW data by the estimated number of households served by landfills in the United States. This resulted in a much higher MSW generation rate per household than other estimates, but this MSW generation rate is consistent with the MSW acceptance rates used in the cost model. Nevertheless, these MSW generation rates per household probably result in overestimates of annualized enterprise costs per household served by affected landfills.

TABLE 8-14. ANNUALIZED ENTERPRISE CONTROL COST PER HOUSEHOLD FOR AFFECTED EXISTING LANDFILLS

	Stringency Level (Mg NMOC/yr)		
	25	100	250
National annualized cost per household ($/Household)	**5.18**	**4.90**	**4.16**
Distribution of affected landfills by annualized cost per household ($/Household)			
≤ 3.50	**313** (21)	**190** (27)	**108** (36)
3.50 to 7.00	**336** (22)	**164** (23)	**75** (25)
7.00 to 15.00	**407** (27)	**184** (26)	**85** (28)
15.00 to 30.00	**216** (14)	**87** (12)	**15** (5)
> 30.00	**249** (16)	**73** (10)	**19** (6)
Total	**1,521** (100)	**698** (100)	**302** (100)

Note: Numbers in parentheses are percentages. Costs for publicly owned landfills are annualized at 4 percent over the control period. Costs for privately owned landfills are annualized at 8 percent from 1992 to the year of closure. Details may not add to totals due to rounding. Excludes closed landfills.

the affected landfills experience annualized costs per household of $3.50 or less and 61% incur costs of $7.00 per household or less.

A measure of the potential cost to society of complying with the regulatory alternatives is the net present value of social costs. This measure, shown in Table 8-15, is computed by first annualizing capital costs and then discounting the flow of capital and operating costs to arrive at a measure of the present value of the costs that will be incurred throughout the control periods for the various landfills. A net present value measure of costs is the appropriate way to compare costs between landfills since most landfills will begin and end controls at different times.

As noted in Section 8.3, computing the net present value of social costs involves a two-stage process. First, the capital costs, which are incurred in discrete "lumps" periodically throughout the control period, are annualized over the control period using a 10% rate. Then the resulting stream of annualized capital costs and the stream of annual operating costs are discounted using a 3% discount rate. These costs are combined to yield the total net present value (NPV) of social costs incurred by each affected landfill. The maximum NPV of social costs incurred by any landfill is $140 million under the 25 Mg stringency level, $112 million under the 100 Mg stringency level, and $75 million under the 250 stringency level.

When summed across all affected landfills under each stringency level, the national total NPV of social costs ranges from $3.92 billion under the 250 Mg stringency level to $11.65 billion under the 25 Mg stringency level (see Table 8-15). While more landfills are affected under the more stringent 25 Mg level than under the other two stringency levels, a larger proportion of affected landfills incurs relatively lower NPV of social costs ($3 million or less) under the 25 Mg level than under the 100 Mg level or the 250 Mg level. The mean NPV of social costs per affected landfill under the 250 Mg stringency, $10.1 million, exceeds the mean NPV of social costs for the other two stringency levels.

Annualizing the net present value of social costs provides another measure of the cost to society of the regulatory alternatives under consideration. In this situation we annualized the net present value of the social cost of each affected landfill over the years from 1992 to the

TABLE 8-15. NET PRESENT VALUE OF SOCIAL COSTS FOR AFFECTED CLOSED AND EXISTING LANDFILLS

Net Present Value	Stringency Levels (Mg NMOC/yr) 25	100	250
National social costs (10^6)			
Capital	6,438	4,326	2,403
Operating	5,213	2,831	1,514
Total	11,651	7,157	3,917
Average total social cost per affected landfill (10^6)	6.18	8.39	10.1
Distribution of affected landfills by net present value of social costs (10^6)			
≤ 0.5	31	29	7
	(2)	(3)	(2)
0.5 to 1.0	97	24	7
	(5)	(3)	(2)
1.0 to 3.0	654	206	61
	(35)	(24)	(16)
3.0 to 5.0	421	189	92
	(22)	(22)	(24)
5.0 to 10.0	464	261	137
	(25)	(31)	(35)
>10.0	217	144	82
	(11)	(17)	(21)
Total	1,884	853	386
	(100)	(100)	(100)

Note: Numbers in parentheses are percentages. Net present value of social cost is computed using a two-step discounting procedure. First, capital costs are annualized at 10 percent over the control period. Then, present values are computed by discounting annual operating costs and annualized capital costs at 3 percent. Details may not add to totals due to rounding.

end of the landfill's control period using a 3% discount rate, and then we summed these individual annualized values to get the total annualized social cost. The resulting total annualized social cost for affected closed and existing landfills for each stringency level is:

- $416 million for the 25 Mg stringency level
- $297 million for the 100 Mg stringency level
- $150 million for the 250 Mg stringency level.

Thus, the annualized social cost of the 100 Mg stringency level is almost twice the annualized social cost of the 250 Mg stringency level. The annualized social cost of the 25 Mg stringency level is 40% higher than the annualized social cost for the 100 Mg stringency level.

8.4.1.2 Energy Recovery Option. As discussed in Section 8.3, it will be more economical for some landfills to reduce emissions by using flares, while for others it will be more economical to use an energy recovery technique. While energy recovery is more costly, especially in terms of initial capital investment, it also will bring in some revenue from the sale of the purified landfill gas or the energy produced from various uses of this gas. In considering the energy recovery options, we omit the landfills that would actually profit from energy recovery according to the model in Chapter 7, because we assume these landfills would initiate the use of energy recovery even in the absence of EPA emissions control regulations. We therefore conclude that neither the emissions reductions nor the costs of emissions control with energy recovery at these landfills should be attributed to the regulatory alternatives under consideration. So assessing the impacts of these regulatory alternatives involves studying only those landfills that would experience positive costs using the least costly control option.

When we omit all landfills that would find energy recovery profitable (that is, landfills where the revenue from energy recovery exceeds the energy recovery costs), the number of affected landfills at each potential level of stringency is considerably smaller. As Table F-1 in Appendix F shows, the number of affected landfills falls from 1884 to 1024, a decrease of 46% under the most stringent regulatory alternative (i.e., 25 Mg of NMOC/yr). At the 100 Mg stringency level, the number of affected landfills

falls by 62%, from 853 to 325. Finally, at the least stringent 250 Mg level, the number of affected landfills falls by 80%, from 386 to only 77.

Table F-1 also shows the number of privately owned affected landfills under the energy recovery option. As described above, privately owned landfills may have the greatest difficulty paying for the emissions controls, because all their costs must be recaptured through increased user fees during the period when the landfill is still actively accepting MSW. The number of privately owned affected landfills varies from 27 under the least stringent 250 Mg cutoff to 68 under the 100 Mg stringency level, and 215 under the 25 Mg stringency level. From 10 to 29 of the privately owned landfills will close by 1992 and therefore are expected to have no way of recapturing the costs of compliance.

As described above, landfills must use emissions controls during a control period that will vary in length from landfill to landfill, extending beyond the closure of the landfill. Table F-2 depicts the length of control period, while F-3 shows the length of control period prior to closure. Although the control period may be as long as 130 years under the 250 Mg stringency level, 235 years under the 100 Mg stringency level, and 277 years under the 25 Mg stringency level, the average length of the control period is much shorter. The average control period for affected landfills under the 250 Mg stringency level is 36 years, while it is 51 years under the 100 Mg stringency level, and it is 70 years under the 25 Mg stringency level. Also, as shown in the frequency distribution of affected landfills by length of control period, the proportion of affected landfills with control periods less than, for example, 50 years, is roughly two-thirds under the 250 Mg and 100 Mg stringency levels, but is only 43% under the 25 Mg stringency level.

The shorter the time between the imposition of controls and a landfill's closure, the more difficult it will be for the landfill to recover a given amount of compliance costs by increasing user fees at the landfill. This problem, of course, is particularly serious for landfills which are already closed, but it may also affect landfills with a fairly short period of time (for example, only 5 or 10 years) between the start of the controls and the landfill's closure. Table F-3 shows the length of the control period prior to closure for existing landfills under the energy recovery

option. While some landfills have as much as 177 years of operating life under the 25 Mg stringency level, the average length of control period prior to closure for that stringency level is about 21 years. For the less stringent levels, the average operating lives are even shorter--14.5 years for the 100 Mg stringency level and less than 9 years for the 250 Mg stringency level. A larger share of the affected landfills will have shorter control periods before closure at the less stringent 250 Mg and 100 Mg levels of control than at the most stringent 25 Mg level. At the 250 Mg stringency level, 81% have 10 years or less of controls prior to closure, while 63% have ten years or less prior to closure at the 100 Mg stringency level, and 41% have 10 years or less prior to closure at the 25 Mg stringency level.

To measure the impacts of the regulatory alternatives under consideration on the owners of landfills, we use the net present value (NPV) of enterprise costs. These costs include both capital investments and operating costs, less revenues from energy recovery for those landfills that choose the energy recovery option. Table F-4 shows these costs, along with a frequency distribution of landfills by NPV of enterprise costs. We assume that the landfill will choose the control option that minimizes its costs of control. To determine which option a particular landfill will select, we discount the capital and operating costs incurred over time to compute a NPV of each. For publicly owned landfills, we use a 4% discount rate, while for privately owned landfills we use an 8% discount rate. The NPV of enterprise costs for the flare control option for each landfill is compared with the NPV of enterprise costs for the energy recovery option minus the revenue from the energy recovery activity.

Allowing landfills to employ an energy recovery control option has two overall effects on the impacts of the regulation. First, fewer landfills are affected, because we assume that any landfill for which the energy recovery option is profitable would have instituted such a system in the absence of any EPA emissions regulation. Thus, we can attribute neither the emissions reductions nor the costs of installing and operating energy recovery equipment to the regulatory alternatives under consideration. Second, the remaining landfills incur lower enterprise costs, both in the aggregate and on average. As just noted, the number of landfills affected

by the regulation falls for each stringency level. As a result, we would expect aggregate NPV of enterprise costs to be lower, even if the average NPV of enterprise costs per landfill did not decrease. In fact, however, the average NPV of enterprise costs per landfill does decrease, falling 54% to 68% when we allow landfills to choose the least costly control option (see Table F-4). At the 100 Mg stringency level, for example, the average NPV of enterprise costs per landfill under the flare option is $4.26 million. When the landfills are allowed to choose their least costly control option, the average landfill now only incurs an NPV of enterprise costs of $1.39 million. As a result of these combined trends, the aggregate NPV of enterprise cost falls by 75% and 93%, depending on the stringency level. The frequency distribution of affected landfills by NPV of enterprise costs is even more skewed toward the lower cost categories under the energy recovery option than under the flare option. At the 25 Mg stringency level, for example, 71% of landfills incur NPV of enterprise costs less than $3 million under the flare option, while 93% of landfills incur NPV of enterprise costs less than $3 million under the energy recovery option.

Annualized enterprise cost is another measure of the impacts of enterprise costs on landfill owners. This is computed for publicly owned landfills by annualizing the NPV of enterprise costs for each landfill using a 4% interest rate over the period during which controls are in place for that landfill. Costs for privately owned landfills are computed by annualizing the NPV of enterprise costs for each landfill using an 8% interest rate over the period from 1992 through the year when the landfill closes.

Table F-5 displays the annualized enterprise costs per Mg of MSW for landfills having positive energy recovery costs. This is computed by dividing the NPV of enterprise costs by the reported quantity of waste accepted in 1986. The national annualized cost per Mg of MSW accepted is computed by summing annualized enterprise cost for all the affected landfills under each stringency level, and then dividing by the sum of the reported quantities of waste accepted by all affected landfills in 1986. These quantities range from $1.43/Mg of MSW accepted at the 250 Mg stringency level to $2.66/Mg of MSW at the 100 Mg stringency level. The

national annualized cost per Mg of MSW for the 25 Mg stringency level falls between those values, at $1.64/Mg of MSW accepted. Although these costs are low, they are about two to three times higher than the national annualized enterprise costs per Mg of MSW under the flare option (see Table 8-11). This occurs because many of the affected landfills with low enterprise costs per Mg of MSW under the flare option will make a profit from energy recovery. So these low unit cost landfills are omitted from the group of affected landfills under the energy recovery option.

Table F-5 also shows a frequency distribution of affected landfills by annualized cost per Mg of MSW. The proportion of affected landfills experiencing annualized costs exceeding $3.00 per Mg is 43% under both the 25 Mg stringency level and the 100 Mg stringency level; the maximum annualized cost incurred at the 25 Mg level is $57.15 per Mg, while the maximum is $25.42 per Mg at the 100 Mg level. At the 250 Mg stringency level, the proportion of landfills with annualized costs of $3.00 per Mg or more falls to 24%, and the maximum annualized cost is $8.39.

We measure the impacts of the §111(d) regulatory alternatives under consideration on the users of affected landfills with the annualized enterprise cost per household. This is computed by dividing the annualized enterprise cost by the estimated number of households (based on an average waste generation rate per household) served by the landfill. The national annualized cost per household, shown at the top of Table F-6, is computed by summing the annualized enterprise costs for each affected landfill at each stringency level, and then dividing by the sum of the estimated number of households served by all the affected landfills at that stringency level. The national annualized cost per household varies from $8.33 per household at the 250 Mg stringency level, to $9.50 at the 25 Mg stringency level, to $15.47 at the 100 Mg stringency level. As was the case for annualized costs per Mg of MSW, national annualized household costs under the energy recovery option are much higher than the annualized household costs under the flare option, because many of the low household cost landfills are not affected by the regulatory alternatives under the assumptions of the energy recovery option.

The frequency distribution of affected landfills by annualized cost per household suggests that the 821 affected landfills at the 25 Mg

stringency level incur annualized costs per household that are more concentrated at the lower values ($7.00 per household or less) than the costs incurred by the 252 affected landfills at the 100 Mg level. The national average cost per household at the 100 Mg stringency level is about $15, but one-quarter of affected landfills at this level incur annualized costs of $30 per household or more.

The net present value of social costs in Table F-7 measures the potential impacts of the stringency levels under consideration on society. The capital costs of compliance are annualized at a 10% rate, then the resulting stream of annualized capital costs plus operating costs are discounted at a 3% rate to determine the net present value of these costs. The NPV of revenues from energy recovery then are subtracted from total costs for those landfills that use the energy recovery option. As indicated in Table F-7, the national social cost of the regulatory alternatives ranges from $253 million for the least stringent 250 Mg level of control to $2.96 billion for the most stringent 25 Mg level of control. While aggregate costs are higher at the more stringent levels of control, average social cost per landfill is lower, because more landfills with lower costs are affected. Specifically, the average total social cost per affected landfill is $2.89 million at the 25 Mg stringency level, $2.55 million at the 100 Mg stringency level, and $3.27 million at the 250 Mg level.

To provide another perspective on the social cost of the regulatory alternatives under consideration, we calculated the annualized social cost for the three stringency levels for the energy recovery option. Specifically, we annualized the net present value of social cost for each landfill over the years from 1992 to the end of its control period using a 3% discount rate, and then we summed the individual annualized values to estimate the total annualized social cost. These costs are:

- $124 million for the 25 Mg stringency level
- $68 million for the 100 Mg stringency level
- $19 million for the 250 Mg stringency level.

Note that annualized social cost exceeds $100 million only for the most stringent regulatory alternative under the energy recovery option. Furthermore, these annualized social costs are much lower than the

annualized social cost of the three stringency levels under the flare option. Specifically, the annualized social cost of the 100 Mg stringency level under the energy recovery option ($68 million) is just one-fourth of the annualized social cost of this same stringency level under the flare option ($297 million).

8.4.2 Section 111(b) Standards

The §111(b) Standards apply to landfills constructed and opened after 1992 when the regulation takes effect. In our case, we assume these new landfills are replacing other landfills that closed. Specifically, we assume that every landfill that closes after 1992 is replaced by an identical landfill serving the same area.

8.4.2.1 Flare Option. Of 944 new landfills nationwide, there are 41 affected by the flare option at the 250 Mg stringency level, 104 affected by the flare option at the 100 Mg stringency level, and 247 affected by the flare option at the 25 Mg stringency level. Tables 8-16 through 8-18 provide information on these affected landfills.

Table 8-16 shows the number of affected new landfills, along with the number of such landfills which are privately owned. As with the closed/existing landfills, privately owned new landfills will need to recapture the costs of compliance with the regulation while they are still accepting MSW. At the 25 Mg level of stringency, 51 of the affected landfills are privately owned, 24 are privately owned at the 100 Mg stringency level, while 14 are privately owned at the 250 Mg stringency level. Table 8-16 also shows a frequency distribution of affected new landfills by design capacity. At the most stringent 25 Mg cutoff level the majority of affected landfills have less than 5 million Mg of capacity, while at the less stringent levels of control the majority are larger.

Table 8-17 depicts the length of control periods for affected new landfills. Again, the landfills must operate the emissions controls for as long as their emissions exceed the selected cutoff level. The year when controls must begin varies from landfill to landfill; the length of time during which controls must be operated also varies from landfill to landfill, and so, therefore, does the date when controls may be removed. While some landfills must keep controls in place for as long as 124 years,

TABLE 8-16. SUMMARY INFORMATION FOR AFFECTED NEW LANDFILLS

	Stringency Levels (Mg NMOC/yr)		
	25	100	250
Number of affected landfills	247	104	41
(Percent of total new landfills)	(26)	(11)	(4)
Distribution of affected landfills by design capacity (10^6 Mg)			
≤ 1	58	0	0
	(23)	(0)	(0)
1 to 5	121	46	10
	(49)	(44)	(24)
5 to 10	29	22	14
	(12)	(21)	(34)
> 10	39	36	17
	(16)	(35)	(41)
Total	247	104	41
	(100)	(100)	(100)
Privately owned affected landfills	51	24	14
(Percent of affected landfills)	(21)	(23)	(35)

Note: The numbers in parentheses are percentages. Details may not add to totals due to rounding.

TABLE 8-17. LENGTH OF CONTROL PERIOD FOR AFFECTED NEW LANDFILLS

	Stringency Levels (Mg NMOC/yr)		
	25	100	250
Average length of control period (years)	74.4	59.6	59.1
Distribution of affected landfills by length of control period (years)			
≤ 25	31 (13)	17 (16)	9 (22)
26 to 50	63 (26)	41 (39)	10 (24)
51 to 100	61 (25)	22 (21)	17 (41)
101 to 150	92 (37)	24 (23)	5 (12)
Total	247 (100.0)	104 (100.0)	41 (100.0)

Note: Numbers in parentheses are percentages. Details may not add to totals due to rounding.

TABLE 8-18. LENGTH OF CONTROL PERIOD PRIOR TO CLOSURE FOR AFFECTED NEW LANDFILLS

	Stringency Levels (Mg NMOC/yr)		
	25	100	250
Average length of control period prior to closure (years)	14.3	13.3	13.3
Distribution of affected landfills by length of control period prior to closure (years)			
≤ 5	36 (14)	17 (16)	7 (17)
6 to 10	32 (13)	5 (5)	10 (24)
11 to 20	152 (62)	75 (72)	17 (42)
21 to 50	27 (11)	7 (7)	7 (17)
Total	247 (100)	104 (100)	41 (100)

Note: Numbers in parentheses are percentages. Details may not add to totals due to rounding.

the average length of control period is about 60 years for the 250 Mg and 100 Mg stringency levels, and 74 years for the 25 Mg stringency level. Table 8-17 also shows that the more stringent the level of control, the higher the proportion of landfills that will incur long periods of control.

Table 8-18 shows the average length of control period prior to closure for affected new landfills, and a frequency distribution of affected landfills by length of control prior to closure. In general, most affected new landfills need not begin controlling emissions until fairly close to their closure date. The average length of time between beginning controls and closure is 13 or 14 years. At the 25 Mg stringency level, 14% of affected landfills will have only 5 years or less of controls before closure, while 16% will have 5 years or less at the 100 Mg stringency level. Finally, 17% will have 5 years or less at the 250 Mg level.

Table 8-19 provides another measure of the severity of impacts on landfill owners from the regulatory alternatives under consideration. It describes the net present value of enterprise costs for affected new landfills. As discussed above, the streams of capital and operating costs incurred by the landfill owners over time are discounted to their present value in order to compare one landfill's costs to another's. To reflect the differences in the cost of capital for private and public landfill owners, different discount rates are used in the discounting process: costs for publicly owned landfills are discounted using a 4% rate, while the costs for privately owned landfills are discounted using an 8% rate. The net present value of capital costs and the net present value of operating costs are summed for each landfill, which yields the total net present value of enterprise costs. These costs are summed across landfills to estimate the aggregate (nationwide) net present value of enterprise costs.

Table 8-19 shows that the 247 new landfills affected by the 25 Mg level of control have total enterprise costs of $641 million, while the 104 new landfills affected by the 100 Mg level of stringency have an aggregate net present value of enterprise costs of $407 million, and the 41 new landfills affected by the 250 Mg stringency level have aggregate net present value of enterprise costs of $249 million. Although some landfills have a NPV of enterprise costs as high as $22 million at each stringency level, the average NPV enterprise costs per landfill are much lower. While the

TABLE 8-19. NET PRESENT VALUE OF ENTERPRISE COSTS FOR AFFECTED NEW LANDFILLS

Net Present Value	Stringency Levels (Mg NMOC/yr) 25	100	250
National enterprise costs (10^6)			
Capital	245	177	117
Operating	396	230	132
Total	641	407	249
Average total enterprise cost per affected landfill (10^6)	2.60	3.92	6.07
Distribution of affected landfills by net present value of enterprise costs (10^6)			
≤ 0.5	39	7	2
	(16)	(7)	(5)
0.5 to 1.0	41	10	2
	(17)	(10)	(5)
1.0 to 3.0	111	53	23
	(45)	(51)	(56)
3.0 to 5.0	36	14	2
	(14)	(13)	(5)
>5.0	20	20	12
	(8)	(19)	(29)
Total	247	104	41
	(100)	(100)	(100)

Note: Numbers in parentheses are percentages. Net present value of enterprise costs is calculated using a 4 percent discount rate for publicly owned landfills and an 8 percent discount rate for privately owned landfills. Details may not add to totals due to rounding.

aggregate NPV enterprise costs are highest at the 25 Mg stringency level, the average NPV enterprise cost per facility for this level, $2.60 million, is lower than for the other two stringency levels, because so many more landfills with lower costs are affected by the 25 Mg stringency level. At the 100 Mg stringency level, the average NPV enterprise cost per facility is $3.92 million, while the average NPV enterprise cost per facility is $6.07 million at the 250 Mg stringency level.

The frequency distribution of affected new landfills by NPV of enterprise costs in Table 8-19 indicates that a higher proportion of affected landfills under the more stringent control alternatives experience a relatively low NPV of enterprise costs. For example, under the 25 Mg stringency level, one-third of affected facilities have a NPV of enterprise costs of $1 million or less. Under the 100 Mg stringency level, one-sixth have a NPV of enterprise costs of $1 million or less, and only 10% have a NPV of enterprise costs of $1 million or less under the 250 Mg stringency level.

Annualizing enterprise costs is another way of using these costs to assess impacts on landfill owners. The NPVs of enterprise costs for publicly owned landfills are annualized using a 4% rate of interest over the period of time during which the controls will be in place. For privately owned landfills, we annualize enterprise costs using an 8% rate of interest during the active operating life of the landfill, since privately owned landfills will not be able to recapture the costs of compliance after they close. We then divide these annualized enterprise costs by the reported quantity of waste that the landfills accepted in 1986.

The first line in Table 8-20 shows the national annualized enterprise cost per Mg of MSW accepted by affected new landfills for each stringency level. This is computed by summing the annualized enterprise cost for all affected landfills at a stringency level, and then dividing by the total MSW accepted by all those landfills. The national annualized cost per Mg of MSW accepted is less than $1.00 per Mg for all stringency levels. At the 250 Mg stringency level, the national cost is $0.46 per Mg. As the stringency increases to the 100 Mg level, the national annualized cost increases to $0.48 per Mg of MSW. At the most stringent 25 Mg cutoff level, the national annualized cost rises to $0.60 per Mg of MSW accepted.

TABLE 8-20. ANNUALIZED ENTERPRISE CONTROL COST PER Mg OF MSW FOR AFFECTED NEW LANDFILLS

	Stringency Level (Mg NMOC/yr)		
	25	100	250
National annualized cost per Mg MSW ($/Mg MSW)	**0.60**	**0.48**	**0.46**
Distribution of affected landfills by annualized cost per Mg MSW ($/Mg MSW)			
≤ 0.25	**10** (4)	**12** (12)	**5** (12)
0.25 to 0.50	**41** (17)	**31** (30)	**14** (34)
0.50 to 1.00	**77** (31)	**24** (23)	**12** (29)
1.00 to 3.00	**75** (30)	**37** (36)	**10** (24)
> 3.00	**44** (18)	**0** (0)	**0** (0)
Total	**247** (100)	**104** (100)	**41** (100)

Note: Numbers in parentheses are percentages. Costs for publicly owned landfills are annualized at 4 percent over the control period. Costs for privately owned landfills are annualized at 8 percent over the life of the landfill. Details may not add to totals due to rounding.

Table 8-20 also has a frequency distribution of affected landfills by the annualized enterprise cost per Mg of MSW accepted. This distribution reveals that, the higher the stringency level, the higher the proportion of affected landfills incurring annualized costs greater than $1.00 per Mg of MSW accepted. At the least stringent 250 Mg cutoff level, only one-quarter of the 41 affected landfills have costs of $1.00 per Mg or higher, and no affected landfill experiences annualized costs exceeding $1.15 per Mg. At the 100 Mg stringency level, however, over one-third of the 104 affected landfills have annualized costs at least as high as $1.00 per Mg; at this stringency level, the maximum annualized cost is $1.89 per Mg of MSW. Finally, at the most stringent 25 Mg level, almost half of the 247 affected landfills have annualized costs of $1.00 per Mg or higher, and at least two landfills have annualized costs of $5.88 per Mg.

Table 8-21 assesses the potential impact of the regulatory alternatives on the households that will be served by these new landfills based on the annualized enterprise cost per household. We compute the overall annualized enterprise cost per household by summing the annualized enterprise costs for each affected landfill under each stringency level, and then we divide the summed annualized enterprise costs by the estimated number of households served by the affected landfills. The national cost per household varies from $2.69 at the 250 Mg stringency level to $2.78 at the 100 Mg stringency level to $3.48 at the 25 Mg stringency level.

As we found for closed/existing landfills, the 25 Mg stringency level has the highest proportion of affected new landfills incurring relatively high costs per household. At that stringency level, over three-fourths of the 247 affected landfills incur costs of $3.00 per household or more. At the 250 Mg stringency level, the proportion of landfills incurring costs of more than $3.00 per household falls to about one-half. At the 100 Mg stringency level, the proportion of affected landfills incurring costs per household as high as $3.00 is lowest of all—only 7% of the 104 affected landfills have costs that high.

Another way of assessing the possible impact of the regulatory alternatives under consideration is to examine the net present value (NPV) of social costs resulting from each possible stringency level (see Table 8-22). As with the NPV of enterprise costs, the aggregate total NPV of

TABLE 8-21. ANNUALIZED ENTERPRISE CONTROL COST PER HOUSEHOLD FOR AFFECTED NEW LANDFILLS

	Stringency Level (Mg NMOC/yr)		
	25	100	250
National annualized cost per household ($/Household)	3.48	2.78	2.69
Distribution of affected landfills by annualized cost per household ($/Household)			
≤ 0.75	2 (1)	15 (14)	0 (0)
0.75 to 1.50	7 (3)	29 (28)	7 (17)
1.50 to 3.00	44 (18)	53 (51)	12 (29)
3.00 to 10.00	121 (49)	7 (7)	22 (54)
> 10.00	73 (30)	0 (0)	0 (0)
Total	247 (100)	104 (100)	41 (100)

Note: Numbers in parentheses are percentages. Costs for publicly owned landfills are annualized at 4 percent over the control period. Costs for privately owned landfills are annualized at 8 percent over the life of the landfill. Details may not add to totals due to rounding.

TABLE 8-22. NET PRESENT VALUE OF SOCIAL COSTS FOR AFFECTED NEW LANDFILLS

Net Present Value	Stringency Levels (Mg NMOC/yr) 25	100	250
National social costs (10^6)			
Capital	788	548	362
Operating	614	348	200
Total	1,403	896	562
Average total social cost per affected landfill (10^6)	5.68	8.63	13.7
Distribution of affected landfills by net present value of social costs (10^6)			
≤ 0.5	7	0	0
	(3)	(0)	(0)
0.5 to 1.0	17	0	0
	(7)	(0)	(0)
1.0 to 3.0	92	39	7
	(37)	(37)	(17)
3.0 to 5.0	44	7	7
	(18)	(7)	(17)
5.0 to 10.0	65	36	15
	(26)	(35)	(37)
> 10.0	22	22	12
	(9)	(21)	(29)
Total	247	104	41
	(100.0)	(100)	(100)

Note: Numbers in parentheses are percentages. Net present value of social cost is computed using a two-step discounting procedure. First, capital costs are annualized at 10 percent over the control period. Then, present values are computed by discounting annual operating costs and annualized capital costs at 3 percent. Details may not add to totals due to rounding

social costs increases as the level of stringency increases. At the most stringent 25 Mg cutoff level, the aggregate total NPV of social costs, $1.4 billion, is more than twice the aggregate total NPV of social costs at the 250 Mg level, $562 million. The aggregate total NPV of social costs at the 100 Mg level, $896 million, lies between the cost of the other stringency levels. Also following the pattern demonstrated by the enterprise costs, the number of affected landfills increases substantially as the stringency level increases, and the average NPV of social costs per landfill decreases as the level of stringency increases. While some landfills have NPV of social costs as high as $51 million, the average NPV of social costs per affected landfill ranges from $13.7 million at the 250 Mg stringency level, to $8.63 million at the 100 Mg stringency level, to $5.68 million at the 25 Mg stringency level. Finally, the frequency distribution in Table 8-22 shows, in a different manner than the averages, that the smaller number of affected landfills at the lower stringency levels have a higher NPV of social costs per landfill.

Our last measure of the cost to society of the §111(b) regulatory alternatives under consideration is the annualized net present value of social costs. As explained above, we annualized the net present value of the social cost for each affected landfill over the years from 1992 to the end of the landfill's control period using a 3% discount rate, and then we summed these individual annualized values to get the total annualized social cost. The resulting total annualized social cost for affected new landfills for each stringency level is:

- $45 million for the 25 Mg stringency level
- $30 million for the 100 Mg stringency level
- $19 million for the 250 Mg stringency level.

As expected, the least stringent regulatory alternative (the 250 Mg stringency level) has the lowest annualized social cost, while the most stringent regulatory alternative (the 25 Mg stringency level) has the highest annualized social cost.

Up to this point, we have assumed that the §111(b) regulatory alternatives under consideration will not affect the quantity of MSW going to new landfills. Actually, landfill emissions controls will increase the cost of landfilling relative to other MSW disposal options (i.e., incineration),

which will provide an incentive for some substitution among disposal technologies. In other words, increases in landfill costs attributable to §111(b) controls will cause a shift in MSW flows away from landfills and towards MWCs. However, EPA is also considering other regulations affecting both landfills and MWCs, as explained in Section 8.2. The net effect of all these regulations on MSW flows is not clear.

To help determine the possible effects on MSW flows of various EPA regulations under consideration, developed an econometric model of the actual choices made by communities between 1980 and 1986 with respect to building either a new landfill or a new MWC.[141] This model estimates the share of MSW going to landfills and MWCs based on disposal costs and the socioeconomic characteristics of communities. By adding the estimated control costs associated with various landfill and MWC regulations to landfill and MWC disposal costs, respectively, the model predicts changes in MSW flows attributable to the regulations.

Table 8-23 presents the results of applying the Bentley/Spitz model incrementally to three EPA regulations: the Subtitle D controls under the Resource Conservation and Recovery Act, the CAA §111(b) controls applying to MWCs, and the CAA §111(b) controls applying to landfills. Under baseline conditions, about 72% of MSW goes to landfills. In other words, the choices that communities make regarding building new MWCs and landfills result in 72% of their MSW going to landfills and 28% going to MWCs in the absence of any new EPA regulations. The Subtitle D controls will increase the cost of landfilling, which will cause more communities to choose the MWC disposal technology. However, the CAA §111(b) controls under consideration for MWCs will substantially increase the costs of this disposal technology, which will result in a large shift in MSW flows towards landfills according to the Bentley/Spitz model. Finally, the CAA §111(b) controls under consideration for landfills will increase landfilling disposal costs slightly, so these controls will only result in a very small shift in MSW flows towards MWCs.*

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*As indicated in Table 8-20, the annualized enterprise control cost per Mg of MSW for affected new landfills is $0.48 under the 100 Mg stringency level. In contrast, the annualized enterprise control cost per Mg of MSW for affected new MWCs is $9.65 for Regulatory Alternative IV under Scenario III.[142] This supports the conclusion that the impact of the landfill emissions controls on MSW flows will be much smaller than the impact of the MWC emissions controls.

TABLE 8-23. MSW TONNAGE SHARES OF MUNICIPAL WASTE COMBUSTORS (MWCS) AND LANDFILLS WITHOUT AND WITH VARIOUS EPA REGULATIONS

	MSW Tonnage Shares		
	MWCs	Landfills	Total
Baseline	27.75%	72.25%	100%
Baseline Plus Subtitle D Control Costs*	30.66%	69.34%	100%
Baseline Plus Subtitle D and MWC Emissions Control Costs**	21.24%	78.55%	100%
Baseline Plus Subtitle D, MWC and Landfills Emissions Control Costs***	21.61%	78.39%	100%

*Estimates of Subtitle D Control costs taken from the RIA[143].

**Estimates of MWC emissions control costs are based on Regulatory Alternative IV under Scenario III[144].

***Landfills emissions control costs are based on the 100 Mg stringency level.

Overall, the three regulations will increase MSW flows to landfills about 6 percentage points (i.e., from 72% to 78%). These results suggest that some increase in MSW acceptance rates at new landfills is appropriate for estimating the costs of the §111(b) regulatory alternatives under consideration for landfills. However, the three assumptions (discussed in Section 8.3) producing high MSW acceptance rates in the costing model in Chapter 7 probably still lead to overestimates of the costs of these regulatory alternatives.

8.4.2.2 Energy Recovery Option. Under the energy recovery option, the landfill owners are allowed to either combust their emissions or control them as part of energy recovery, depending upon which approach is least costly for them. Undoubtedly, some landfills will find energy recovery not only less costly than flares, but actually profitable. We assume that the owners of such landfills would install energy recovery systems even in the absence of the emissions control regulation. Therefore, we do not attribute either the emissions reductions or the costs of these energy recovery systems to the regulatory alternatives under consideration. We limit our analysis, therefore, to those landfills for which the costs of installing and operating emissions controls of either type will be positive. Appendix F has the tables on the affected new landfills having positive energy recovery costs.

By eliminating landfills that profit from energy recovery, the §111(b) regulatory alternatives affect far fewer new landfills. Table F-8 shows that the number of affected new landfills varies from 10 under the least stringent 250 Mg level of control, to 39 under the 100 Mg stringency level, and 140 under the 25 Mg stringency level. Additionally, the frequency distribution of affected new landfills by design capacity reveals that no small landfills (1 million Mg or less) are affected by the 100 Mg and 250 Mg stringency levels under the energy control option. As discussed above, privately owned landfills may have less flexibility in paying for emissions controls, because they must recapture the costs of these controls through increased user fees while the landfill is still accepting MSW. Under the 250 Mg and 100 Mg stringency levels, none of the affected landfills are privately owned. Under the 25 Mg stringency level, however, there are 34 privately owned landfills, which is almost one-quarter of the affected new landfills.

Table F-9 shows the length of the control period for affected new landfills with positive energy recovery costs. The average length of the control period ranges from 56 years for the 100 Mg stringency level to 75 years for the 250 Mg stringency level. The average length of the control period for the 25 Mg and 100 Mg stringency levels is slightly below the average length of the control period for these stringency levels under the flare option (see Table 8-17). However, the average length of the control period under the 250 Mg stringency level increases under the energy recovery option, despite no affected landfills having a control period in excess of 100 years at this stringency level.

Another measure of the potential impacts from the regulatory alternatives is the length of time after controls begin and before closure of the landfill. If the landfill is still accepting MSW, its owners can attempt to increase user fees to recapture some of the costs of compliance. Table F-10 shows the length of control period prior to closure. While there are many fewer affected landfills when landfills that profit from energy recovery are eliminated, the length of control period prior to closure is slightly shorter for the landfills with positive energy recovery costs. Comparing Table F-10 with Table 8-18 reveals that the landfills with positive energy recovery costs have shorter periods of time prior to closure when compared with all affected new landfills under the flare option. Both the average length of control period prior to closure and the distribution of affected landfills by length of control period prior to closure at all three stringency levels demonstrate the difference. Under the flare option, between 14% and 17% of affected new landfills close within five years of implementing emissions controls; alternatively, between 18% and 30% of affected new landfills with positive energy recovery costs close within five years of implementing emissions controls.

To assess the impact of the regulatory alternatives on the owners of affected new landfills under the energy recovery option, we compute the net present value (NPV) of enterprise costs under the flare option and the energy recovery option, omitting landfills that would profit from energy recovery. Then, we assume that the landfill owner will choose the least costly of the control options. To compute the national values at the top of Table F-11, we aggregate the NPV of capital and operating costs for

affected landfills for each stringency level. Then we sum the energy recovery revenues for the landfills that select the energy recovery option for each stringency level. Finally, we calculate the total aggregate NPV of enterprise costs by adding the capital and operating sums and subtracting the revenue sum. At the 250 Mg stringency level, this total equals about $18 million, or an average of $1.83 million for each of the affected new landfills. At the 100 Mg stringency level, the total aggregate NPV of enterprise costs is $63 million, or an average of $1.61 million for each of the affected landfills at that level. Finally, at the 25 Mg stringency level, the total NPV of enterprise costs is $150 million, which averages $1.07 million for each of the affected landfills.

Table F-11 has a frequency distribution of affected new landfills by NPV of enterprise costs. At the 250 Mg stringency level, all the affected landfills experience NPV of enterprise costs between $500,000 and $2.2 million. At the 100 Mg stringency level, all the affected landfills have NPV of enterprise costs between $500,000 and $3.5 million. Finally, NPV of enterprise costs range from below $500,000 to $3.8 million at the 25 Mg stringency level.

Another measure of the impacts of the regulatory alternatives on landfills is the annualized enterprise control cost per Mg of MSW accepted by the landfill. Table F-12 shows the annualized enterprise costs for landfills with positive energy recovery costs when owners are allowed to select the least costly means of achieving emission reductions, either using flares or using energy recovery. At each stringency level, the annualized cost per Mg of MSW is less than $1.00. At the 250 Mg stringency level the overall annualized cost is only $0.59 per Mg. It is $0.92 per Mg at the 100 Mg stringency level, and it is $0.95 per Mg at the 25 Mg stringency level. These national annualized costs per Mg of MSW are between 28% and 92% higher than the national annualized costs per Mg of MSW under the flare option, because many of the low cost per Mg landfills under the flare option are omitted from the affected landfills under the assumptions of the energy recovery option.

The frequency distribution of affected new landfills by annualized enterprise control costs per Mg of MSW in Table F-12 shows that all the affected landfills have annualized costs between $0.50 and $3.00 per Mg for

the 100 Mg and 250 Mg stringency levels. The maximum annualized cost at the 250 Mg stringency level is $1.08 per Mg, and the maximum at the 100 Mg stringency level is $1.42 per Mg. At the 25 Mg stringency level, on the other hand, affected landfills have unit costs ranging from below $0.25 per Mg to $5.30 per Mg. Over one-quarter of the affected landfills under this stringency level have annualized costs per Mg of $3.00 or higher.

To assess the possible impacts of the emissions control alternatives on the households served by affected landfills, we computed the annualized enterprise control costs per household. Table F-13 has these costs for affected landfills with positive energy recovery costs when landfill owners may choose either the flare option or the energy recovery option. At the 250 Mg stringency level, the national annualized cost is $3.41 per household. The annualized cost per household increases to $5.36 at the 100 Mg stringency level, and the annualized cost per household is $5.53 at the 25 Mg stringency level. As was the case for annualized costs per Mg of MSW, national annualized household costs under the energy recovery option are higher than annualized household costs under the flare option for reasons discussed above.

Table F-13 also contains a frequency distribution of affected new landfills by the annualized cost per household. At the 250 Mg stringency level, the 10 affected landfills have annualized costs between $1.50 and $10.00 per household. At the 100 Mg stringency level, the 39 affected landfills have annualized enterprise costs between $3.00 and $10.00 per household. Finally, the 140 affected landfills at the 25 Mg stringency level have annualized enterprise costs ranging from less than $0.75 per household to more than $10.00 per household.

Table F-14 shows another means of measuring the cost of complying with the emissions control regulations under the energy recovery option--the NPV of social costs. The aggregate NPV of social costs falls almost 78% at the 25 Mg stringency level under the energy recovery control option. At the 100 Mg stringency level, the aggregate NPV of social costs falls by 84% under this option, and the aggregate NPV of social costs falls by about 90% at the 250 Mg stringency level compared to the costs under the flare option. This decrease in the aggregate NPV of social costs is largely the result of a reduction in the number of affected landfills. However, the

average total social cost per affected landfill under the energy recovery option is less than half the average total social cost per affected landfill under the flare option for all three stringency levels.

To provide another perspective on the social cost of the §111(b) regulatory alternatives under consideration, we calculated the annualized social cost for the three stringency levels under the energy recovery option. These costs for the affected new landfills under the energy recovery option are:

- $10.5 million for the 25 Mg stringency level
- $4.3 million for the 100 Mg stringency level
- $1.6 million for the 250 Mg stringency level.

These annualized social costs are substantially lower than the annualized social costs under the flare option. For example, the $4.3 million annualized social cost for the 100 Mg stringency level under the energy recovery option is just one-seventh of the $30.2 million annualized social cost for the same stringency level under the flare option.

8.5 ANALYSIS OF EMISSIONS REDUCTIONS AND COST-EFFECTIVENESS

At the same time that we are considering the costs of complying with the §111(d) and 111(b) regulatory alternatives under consideration, we must also consider the cost-effectiveness of these alternatives. In this case cost-effectiveness is measured as the annualized compliance cost per Mg of reduction in the emission of nonmethane organic compounds (NMOCs). We discuss compliance costs for each stringency level and each option in the previous section. In this section, we examine both the emissions reductions and cost-effectiveness of the regulatory alternatives under consideration for both closed/existing and new landfills under each of two control options. We will first examine the emissions reductions and the cost-effectiveness of the flare control option for closed and existing landfills. Then we present the same two measures for these landfills under the energy recovery option. Finally, we examine the emissions reductions and cost-effectiveness of both control options for new landfills.

8.5.1 Section 111(d) Guidelines

As shown in Table 8-6 in Section 8.4, the number of closed and existing landfills affected by the §111(d) Guidelines under the flare control

option ranges from 386 at the 250 Mg stringency level to 853 at the 100 Mg level to 1884 at the 25 Mg stringency level. As explained above, we omit landfills that make a profit from energy recovery when analyzing the impacts of the energy recovery option. So the number of closed and existing landfills affected by the guidelines under the energy recovery option is lower: 77 under the 250 Mg stringency level, 325 under the 100 Mg level, and 1024 under the 25 Mg level.

8.5.1.1 Flare Option. Table 8-24 shows the emissions reductions resulting from the three regulatory alternatives under the flare option. Total undiscounted NMOC emissions reductions range from 24.1 million Mg at the 250 Mg stringency level, to 28.6 million Mg at the 100 Mg stringency level, to 33.2 million Mg at the 25 Mg stringency level. These emissions reductions are spread over the period of time during which the affected landfills are using the flare emission controls. In order to compare emissions reductions with the costs from Section 8.4, we discount the NMOC emissions reductions using a 3% rate of discount. The discounted NMOC emissions reductions range from 9.6 million Mg at the 250 Mg stringency level to 11.2 million Mg at the 100 Mg stringency level to 12.6 million Mg at the 25 Mg stringency level. The average discounted NMOC emission reduction decreases as the stringency level increases, because the number of affected landfills increases faster than the NMOC emissions reductions. Thus, the average NMOC emission reduction per affected landfill is 24,966 Mg at the 250 Mg stringency level, 13,110 Mg at the 100 Mg stringency level, and 6,674 Mg at the 25 Mg stringency level.

We combined these measures of NMOC emissions reductions with the discounted NPV of social costs presented in Table 8-15 to estimate the cost-effectiveness of the flare option for closed and existing landfills (see Table 8-25). At the top of the table is the national cost-effectiveness of each stringency level, computed by dividing the aggregate NPV of total social cost by the total discounted NMOC emissions reduction. The national cost-effectiveness of the flare option at the 250 Mg stringency level is $407 per Mg of NMOC reduced. At the 100 Mg stringency level, the national cost-effectiveness is $640 per Mg of NMOC reduced, and the national cost-effectiveness is $927 per Mg of NMOC reduced at the most stringent 25 Mg level.

TABLE 8-24. NET PRESENT VALUE OF EMISSIONS REDUCTIONS FOR AFFECTED CLOSED AND EXISTING LANDFILLS

Net Present Value	Stringency Levels (Mg NMOC/yr) 25	100	250
Undiscounted NMOC emission reduction (10^6 Mg)	33.2	28.6	24.1
Discounted NMOC emission reduction (10^6 Mg)	12.6	11.2	9.64
Average discounted NMOC emission reduction per affected landfill (Mg)	6,674	13,110	24,966
Distribution of affected landfills by discounted NMOC emission reduction per affected landfill (Mg)			
≤ 1,000	593 (31)	104 (12)	22 (6)
1,000 to 2,000	453 (24)	138 (16)	17 (4)
2,000 to 5,000	425 (23)	228 (27)	43 (11)
5,000 to 10,000	162 (9)	135 (16)	63 (16)
> 10,000	251 (13)	248 (29)	241 (63)
Total	1,884 (100)	853 (100)	386 (100)

Note: Numbers in parentheses are percentages. Net present value of emission reductions is calculated using a 3 percent discount rate. Details may not add to totals due to rounding.

TABLE 8-25. COST EFFECTIVENESS FOR AFFECTED CLOSED AND EXISTING LANDFILLS

	Stringency Level (Mg NMOC/yr)		
	25	100	250
National cost effectiveness ($/Mg NMOC)	927	640	407
Distribution of affected landfills by cost effectiveness ($/Mg NMOC)			
≤ 1,000	382 (20)	433 (51)	295 (76)
1,000 to 2,000	447 (24)	251 (30)	70 (18)
2,000 to 5,000	721 (38)	123 (14)	19 (5)
5,000 to 10,000	269 (14)	24 (3)	2 (1)
> 10,000	65 (4)	22 (2)	0 (0)
Total	1,884 (100)	853 (100)	386 (100)
Incremental cost effectiveness	3,225	2,097	—

Note: Numbers in parentheses are percentages. Cost effectiveness is calculated by dividing the net present value of social cost by the discounted NMOC emission reduction (see Tables 8-15 and 8-24). Details may not add to totals due to rounding.

The frequency distribution of affected landfills by cost-effectiveness demonstrates that as the stringency level decreases, an increasing proportion of landfills has a cost-effectiveness under $1,000 per Mg of NMOC reduced. At the 25 Mg stringency level, only 20% of affected landfills have cost-effectiveness measures that low, while more than half of the affected landfills fall below $1,000 per Mg of NMOC at the 100 Mg stringency level. Finally, three-fourths of the affected landfills have a cost-effectiveness less than $1,000 per Mg of NMOC at the 250 Mg stringency level. At the bottom of the table, incremental cost-effectiveness measures the change in national cost-effectiveness experienced as the stringency level increases first from 250 Mg to 100 Mg, and then from 100 Mg to 25 Mg. As the stringency level increases from 250 Mg to 100 Mg, the incremental cost-effectiveness is $2,097 per Mg of NMOC reduced. Moving from 100 Mg to 25 Mg results in an incremental cost effectiveness of $3,225 per Mg of NMOC reduced.

8.5.1.2 Energy Recovery Option. Table F-15 presents the emissions reductions resulting from the three regulatory alternatives under the energy recovery option. Because so many landfills would find energy recovery profitable, there are far fewer affected landfills under the energy recovery option. Consequently, the total undiscounted NMOC emissions reductions under this option are much less than under the flare option. Specifically, total undiscounted NMOC emissions reductions range from 1.26 million Mg at the 250 Mg stringency level, to 3.06 million Mg at the 100 Mg stringency level, to 5.81 million at the 25 Mg stringency level. These emissions reductions are spread over the period of time during which landfills are operating the emission controls. In order to compare emissions reductions with the costs from Section 8.4, we discount the NMOC emissions reductions using a 3% rate of discount. The discounted NMOC emissions reductions range from 0.59 million Mg at the 250 Mg stringency level to 1.15 million Mg at the 100 Mg stringency level to 2.04 million Mg at the 25 Mg stringency level. The average discounted NMOC emission reduction decreases as the stringency level increases, because the number of affected landfills increases faster than the NMOC emissions reductions. Thus, the average NMOC emission reduction per affected landfill is 7,560 Mg at the 250 Mg stringency level, 3,546 Mg at the 100 Mg stringency level, and 1,993

Mg at the 25 Mg stringency level. The averages are less than one-third of the average NMOC emission reductions under the flare option.

Table F-16 shows the social cost-effectiveness of the energy recovery option. The national cost-effectiveness measures are higher at each level of stringency than the cost-effectiveness of the stringency levels under the flare option, with the greatest increase occurring at the 25 Mg stringency level. The frequency distribution of affected landfills by cost-effectiveness under the energy recovery option shows that the affected landfills are concentrated in the lower cost-effectiveness categories at the less stringent levels of control. As under the flare option, the degree of concentration increases as the level of stringency decreases. At the 25 Mg stringency level, only 15% of affected landfills have a cost-effectiveness under $1,000 per Mg of NMOC reduced. At the 100 Mg level, 58% fall below $1,000 per Mg of NMOC, and 88% fall below $1,000 per Mg of NMOC at the 250 Mg level. Also displaying a similar pattern to the flare option, the incremental cost-effectiveness increases as the level of stringency increases, although the measures of incremental cost-effectiveness are much lower at each level of stringency than under the flare option.

8.5.2 Section 111(b) Standards

New landfills will be regulated under the §111(b) Standards. We present measures of emissions reductions and cost-effectiveness for affected new landfills under each control option in this section.

8.5.2.1 Flare Option. Under the flare control option, the number of affected new landfills ranges from 41 at the 250 Mg stringency level, to 104 at the 100 Mg stringency level, to 247 at the 25 Mg stringency level. Table 8-26 shows the emissions reductions for new landfills under this control option. The first line shows the total undiscounted NMOC emissions reductions at each stringency level. These measures, showing the total emissions reductions achieved throughout the control period for all affected new landfills, ranges from 1.74 million Mg at the 250 Mg stringency level, to 2.33 million Mg at the 100 Mg stringency level, to 2.93 million Mg at the 25 Mg stringency level.

In order to compare emissions reductions between landfills when the emissions reductions occur at different times at different landfills, we

TABLE 8-26. NET PRESENT VALUE OF EMISSIONS REDUCTIONS FOR AFFECTED NEW LANDFILLS

Net Present Value	Stringency Levels (Mg NMOC/yr) 25	100	250
Undiscounted NMOC emission reduction (10^6 Mg)	2.93	2.33	1.74
Discounted NMOC emission reduction (10^6 Mg)	0.99	0.83	0.63
Average discounted NMOC emission reduction per affected landfill (Mg)	4,015	7,983	15,278
Distribution of affected landfills by discounted NMOC emission reduction per affected landfill (Mg)			
≤ 1,000	106 (43)	2 (2)	5 (12)
1,000 to 2,000	39 (16)	15 (14)	2 (5)
2,000 to 5,000	68 (27)	53 (51)	2 (5)
5,000 to 10,000	10 (4)	10 (10)	8 (19)
> 10,000	24 (10)	24 (23)	24 (59)
Total	247 (100)	104 (100)	41 (100)

Note: Numbers in parentheses are percentages. Net present value of emission reductions is calculated using a 3 percent discount rate. Details may not add to totals due to rounding.

discount the NMOC emissions reductions using a 3% rate of discount. This discounted NMOC emission reduction, when summed across all affected landfills, ranges from 0.63 million Mg at the 250 Mg stringency level to 0.83 million Mg at the 100 Mg stringency level and 0.99 million Mg at the 25 Mg stringency level.

The average discounted NMOC emission reduction per affected landfill is much higher at the 250 Mg stringency level than at the 25 Mg stringency level because the number of affected landfills falls faster than discounted NMOC reduction as the stringency level decreases. At the 250 Mg stringency level, the average discounted NMOC emission reduction is 15,278 Mg of NMOC, more than three times the average discounted NMOC emission reduction per landfill at the 25 Mg stringency level (4,015 Mg of NMOC). At the 100 Mg stringency level, the average discounted NMOC emission reduction, 7,983 Mg of NMOC per affected landfill, falls between the average emission reduction values of the other two stringency levels. The frequency distribution of affected new landfills by discounted NMOC emission reduction shows that the proportion of landfills achieving relatively greater NMOC emissions reductions increases as the stringency level decreases.

We can construct cost-effectiveness measures for affected new landfills by combining information about emission reduction with information about the NPV of social costs in Table 8-22. Specifically, we estimate national cost-effectiveness by dividing the total social cost by the total discounted emission reduction for each stringency stringency level. As shown in Table 8-27 this value ranges from $897 per Mg of NMOC reduced at the 250 Mg stringency level, to $1,081 per Mg of NMOC at the 100 Mg level, to $1,416 per Mg of NMOC at the 25 Mg stringency level. The frequency distribution demonstrates that, as with closed/existing landfills, the proportion of affected new landfills having cost-effectiveness measures less than $1000 per Mg of NMOC increases as the degree of stringency decreases. At the 25 Mg stringency level, only 13% of landfills have a cost-effectiveness under $1,000 per Mg of NMOC, while at the 100 Mg stringency level, 44% have a cost-effectiveness of $1,000 per Mg or less. At the 250 Mg stringency level, 59% of affected landfills have a cost-effectiveness under $1,000 per Mg.

The last line of Table 8-27 shows incremental cost-effectiveness--i.e., the change in cost-effectiveness experienced as one moves from the 250 Mg stringency level to the 100 Mg level, and then from the 100 Mg stringency level to the 25 Mg stringency level. As the stringency level increases from 250 Mg to 100 Mg, the incremental cost-effectiveness is $1,648 per Mg of NMOC reduced. The incremental cost-effectiveness of moving from the 100 Mg stringency level to the 25 Mg stringency level is $3,136 per Mg of NMOC reduced.

8.5.2.2 Energy Recovery Option. Table F-17 presents the emissions reductions for affected new landfills with positive energy recovery costs. The undiscounted NMOC emission reduction for each stringency level ranges from 0.25 million Mg of NMOC reduced at the 250 Mg stringency level, to 0.49 million Mg of NMOC reduced at the 100 Mg stringency level, to 0.83 million Mg at the 25 Mg stringency level. The discounted NMOC emissions reductions range from 0.06 million Mg at the 250 Mg stringency level to 0.25 million Mg at the 25 Mg stringency level. As the level of stringency decreases, the average discounted NMOC emission reduction per affected new landfill increases, because the number of affected landfills falls more rapidly than the discounted NMOC emissions reductions. At the 25 Mg stringency level, the average discounted NMOC emission reduction per affected landfill is 1,765 Mg. At the 100 Mg stringency level, the average discounted emission reduction is 3,818 Mg per affected landfill, while the average discounted NMOC emission reduction per affected landfill rises to 6,680 Mg per affected landfill at the 250 Mg stringency level. Again, the smaller number of landfills affected at the 250 Mg stringency level experience greater emissions reductions on average. The frequency distribution of affected landfills by discounted NMOC emission reduction per affected landfill (at the bottom of Table F-17) supports this consideration.

Table F-18 shows the cost-effectiveness of the three stringency levels for the energy recovery control option for affected new landfills. The national cost-effectiveness of each stringency level varies from $891 per Mg of NMOC reduced at the 250 Mg stringency level to $963 per Mg of NMOC reduced at the 100 Mg level, to $1,244 per Mg of NMOC reduced at the 25 Mg stringency level. These national cost-effectiveness measures are lower than the cost-effectiveness of the stringency levels under the flare option.

TABLE 8-27. COST EFFECTIVENESS FOR AFFECTED NEW LANDFILLS

	Stringency Level (Mg NMOC/yr)		
	25	100	250
National cost effectiveness ($/Mg NMOC)	1,416	1,081	897
Distribution of affected landfills by cost effectiveness ($/Mg NMOC)			
≤ 1,000	31 (13)	46 (44)	24 (59)
1,000 to 2,000	68 (27)	39 (38)	7 (17)
2,000 to 5,000	102 (41)	19 (18)	10 (24)
5,000 to 10,000	39 (16)	0 (0)	0 (0)
> 10,000	7 (3)	0 (0)	0 (0)
Total	247 (100)	104 (100)	41 (100)
Incremental cost effectiveness	3,136	1,648	—

Note: Numbers in parentheses are percentages. Cost effectiveness is calculated by dividing the net present value of social cost by the discounted NMOC emission reduction (see Tables 8-22 and 8-26). Details may not add to totals due to rounding.

The frequency distribution in Table F-18 demonstrates that the proportion of affected landfills experiencing a cost-effectiveness of $1,000 per Mg of NMOC reduced or less, increases substantially as the level of stringency decreases. At the 25 Mg stringency level, only one-sixth of affected new landfills have a cost-effectiveness of $1,000 per Mg of NMOC or less, while 38% are below that level of cost-effectiveness at the 100 Mg stringency level. At the 250 Mg stringency level, 70% of the affected new landfills have a cost-effectiveness under $1,000 per Mg of NMOC reduced.

Finally, at the bottom of Table F-18, incremental cost-effectiveness is $870 per Mg of NMOC reduced as the stringency level increases from 250 Mg to 100 Mg. Moving from the 100 Mg stringency level to the 25 Mg stringency level results in an incremental cost-effectiveness of $1,661 per Mg of NMOC reduced. These incremental cost-effectiveness values are about one-half of the corresponding incremental cost-effectiveness values under the flare option.

8.6 ANALYSIS OF DISTRIBUTIONAL IMPACTS

The Regulatory Flexibility Act of 1980 requires federal agencies to determine if regulations will have a "significant economic impact on a substantial number of small entitities." According to EPA guidelines,[145] regulatory impacts are significant if:

- compliance costs are greater than five percent of production costs,
- compliance costs, as a percent of sales, are at least 10 percent higher for small entities than for other entities,
- capital costs of compliance are a significant portion of available capital, or
- the regulation is likely to result in closures of small entities.

The guidelines indicate that a "substantial number" of small entities is "more than 20 percent of these (small entities)." Finally, the EPA generally relies upon Small Business Administration guidelines for identifying "small entities."[146] However, the Regulatory Flexibility Act defines small government jurisdictions as those having fewer than 50,000 people. Since over three-fourths of U.S. landfills are owned by government

agencies, the potential impacts of the regulatory alternatives on small governmental entities are very relevant.

As explained below, the §111(d) Guidelines and 111(b) Standards under consideration will not affect a substantial number of small entities under EPA guidelines. Consequently, regulatory flexibility analyses are not required for these two rulemakings. Nevertheless, this section presents some distributional impacts on households and government jurisdictions of the flare option for the three stringency levels under consideration for the §111(d) Guidelines and 111(b) Standards. These distributional impacts rely on household and governmental data developed by EPA's Office of Solid Waste (OSW) for a landfills rulemaking under Subtitle D of the Resource Conservation and Recovery Act (RCRA).

8.6.1 Section 111(d) Guidelines

As indicated earlier in Table 8-6, the 25, 100, and 250 Mg stringency levels for the §111(d) Guidelines affect only 26%, 12%, and 5%, respectively, of all the closed and existing landfills in the United States in 1992. Since most landfills are small (i.e., 1 million Mg of design capacity or less), while the regulatory alternatives under consideration affect mainly large landfills (i.e., landfills with a design capacity over 1 million Mg), it is very unlikely that any of the stringency levels will affect more than 20 percent of the small landfills.*

To further investigate the impacts of the 25, 100, and 250 Mg stringency levels on small landfills, we analyzed the distribution of affected closed and existing landfills by design capacity relative to the total number of closed and existing landfills in the same size categories. All three stringency levels affect less than 10 percent of the closed and

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* Lacking information on the size of governmental jurisdictions served by most landfills, we assume that small landfills serve small municipalities. This assumption is reasonable for two reasons. First, it is very unlikely that small municipalities will have large landfills, given the high cost of developing and operating large landfills. Second, large municipalities generate large amounts of solid waste, which requires a large amount of disposal capacity. Because of economies of scale in landfill operations and the difficulty of siting landfills, large municipalities will probably not be served by several small landfills. However, some large municipalities may be served by a municipal waste incinerator and a small landfill. In such cases, impacts on small landfills will not necessarily imply impacts on small municipalities.

existing landfills having a design capacity of 1 million Mg or less. While the 100 Mg stringency level affects 12% of the closed and existing landfills in total, it affects less than 4% of the closed and existing small landfills (i.e., landfills with 1 million Mg of design capacity or less). In conclusion, the §111(d) Guidelines do not require a Regulatory Flexibility Analysis, because they do not affect a significant number of small entities.

Although a Regulatory Flexibility Analysis is not required by the §111(d) Guidelines, we examine some distributional impacts of the various stringency levels under consideration. As indicated previously, these distributional impacts rely on household and governmental data developed by EPA's OSW for a landfills rulemaking under Subtitle D of RCRA. These data were available for only a subset of the affected closed and existing landfills for the three stringency levels under consideration for the §111(d) Guidelines.* The specific distributional impacts examined for the subset of affected landfills are:

- population of the service area
- annualized control costs per household
- annualized control costs as a percentage of annual local taxes paid by households
- net present value of capital costs as a percentage of net municipal debt (for publicly owned landfills).

The first measure (i.e., population of the service area) shows the number of people served by the affected landfills. This provides information on the size of the communities affected by the regulatory alternatives under consideration. The second measure reflects the potential annual cost of the controls to the households served by the affected landfills. The third

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*The affected closed and existing landfills for which OSW data are available are generally smaller (in terms of design capacity, refuse in place in 1987, and the amount of MSW received in 1986) than the other affected landfills. In fact, the size difference is statistically significant for the affected landfills under the 25 Mg stringency level according to Student-t tests on design capacity and refuse in place. The size differences between the affected closed and existing landfills for which OSW data are available and the other affected landfills are not statistically significant under the 100 Mg and 250 Mg stringency levels.

measure examines the relative impact of the controls on households, by comparing annual control costs to households' annual local tax "burden." Finally, the fourth measure provides some information on the relative size of the capital costs of the regulatory alternatives under consideration for the affected municipalities.

Table 8-28 shows the population of the service area for the subset of affected closed and existing landfills. Approximately half of the affected landfills serve between 10,000 and 50,000 people under all three stringency levels. In general, as the stringency level increases, more landfills serving smaller communities are affected, as indicated by the changes in the distribution of affected landfills by the service area population. About one-fifth of the affected landfills at the 100 Mg stringency level serve 10,000 people or less, while another one-fifth serve 10,000 to 25,000 people.

The households served by more than two-thirds of the subset of affected closed and existing landfills incur less than $25 per year in control costs under all three stringency levels (see Table 8-29). The households served by 18% of the affected landfills incur more than $50 per year in control costs under the 100 Mg stringency level.* Nevertheless, the national average control cost per household is just $13 for the 100 Mg stringency level.

To further investigate the potential household impacts of the emissions controls under consideration, Table 8-30 shows annualized control costs as a percentage of local taxes paid by households in the service area of the subset of affected closed and existing landfills. The national average control cost as a percentage of local taxes paid by households is under 1.3% for all three stringency levels. Control costs as a percentage of local taxes paid are less than or equal to 1% for households served by 40% of the affected landfills at the 100 Mg stringency level. At the other

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*The number of households in the service areas of these landfills is low compared to the amount of MSW going into the landfills. In other words, the amount of waste going into the landfills in these areas implies a greater number of households based on the typical amount of MSW generated by households. So the relatively high household costs for these affected landfills are a result of overestimated control costs stemming from overestimated MSW acceptance rates and/or underestimated numbers of households served by these landfills.

TABLE 8-28. SERVICE AREA POPULATION FOR A SUBSET OF THE AFFECTED CLOSED AND EXISTING LANDFILLS

	Stringency Levels (Mg NMOC/yr)		
	25	100	250
National average service area population (10^3 people)	79.5	138.9	107.2
Distribution of affected landfills by service area population (10^3 people)			
≤ 10	315 (28)	95 (21)	15 (8)
10 to 25	278 (25)	89 (19)	41 (23)
25 to 50	254 (23)	133 (28)	48 (27)
50 to 150	169 (15)	82 (17)	34 (19)
150 to 500	65 (6)	48 (10)	38 (21)
> 500	27 (2)	24 (5)	5 (3)
Total	1,108 (100)	471 (100)	181 (100)

Note: The numbers in parentheses are percentages. Details may not add to totals due to rounding.

TABLE 8-29. ANNUALIZED ENTERPRISE COSTS PER HOUSEHOLD FOR A SUBSET OF THE AFFECTED CLOSED AND EXISTING LANDFILLS

	Stringency Levels (Mg NMOC/yr)		
	25	100	250
National average annualized cost per household ($/Household)	9.49	12.91	9.46
Distribution of affected landfills by annualized cost per household ($/Household)			
≤ 5	217 (20)	89 (19)	56 (31)
5 to 10	239 (22)	131 (28)	46 (26)
10 to 25	303 (27)	94 (20)	41 (23)
25 to 50	170 (15)	72 (15)	19 (10)
> 50	179 (16)	85 (18)	19 (10)
Total	1,108 (100)	471 (100)	181 (100)

Note: Numbers in parentheses are percentages. Costs for publicly owned closed and existing landfills are annualized at 4 percent over the control period. Costs for privately owned existing landfills are annualized at 8 percent from 1992 to the year of closure. Costs for privately owned closed landfills are annualized at 8 percent over the the control period. Costs for Details may not add to totals due to rounding.

TABLE 8-30. ANNUALIZED ENTERPRISE COST AS A PERCENTAGE OF LOCAL TAXES PAID BY HOUSEHOLDS IN THE SERVICE AREA FOR A SUBSET OF THE AFFECTED CLOSED AND EXISTING ~~AFFECTED~~ LANDFILLS

	Stringency Levels (Mg NMOC/yr)		
	25	100	250
National average annualized enterprise cost as a percent of taxes paid by households (%)	0.9	1.2	1.0
Distribution of affected landfills by average annualized cost as a percent of taxes paid by households (%)			
≤ 1	452 (41)	189 (40)	77 (43)
1 to 2.5	341 (31)	142 (30)	68 (38)
2.5 to 10	208 (19)	89 (19)	29 (16)
> 10	107 (10)	51 (11)	7 (4)
Total	1,108 (100)	471 (100)	181 (100)

Note: Numbers in parentheses are percentages. Costs for publicly owned closed and existing landfills are annualized at 4 percent over the control period. Costs for privately owned existing landfills are annualized at 8 percent from 1992 to the year of closure. Costs for privately owned closed landfills are annualized at 8 percent over the the control period. Details may not add to totals due to rounding.

extreme, control costs exceed 10% of local taxes paid for households served by one-ninth of the affected landfills under this same stringency level.*

As a final measure of the distributional impact of the §111(d) regulatory alternatives under consideration, Table 8-31 examines the net present value of capital costs as a percentage of net municipal debt for a subset of affected publicly owned closed and existing landfills. Overall, the capital costs of the three stringency levels under consideration represent less than 2.5% of the net debt of municipalities served by publicly owned closed and existing landfills. Capital costs are less than or equal to 5% of municipal debt for the municipalities served by over six-tenths of affected landfills under the 100 Mg stringency level. However, capital costs are more than double the net municipal debt for the municipalities served by about 2% of the affected landfills at this stringency level.†

In conclusion, the distributional impacts of the §111(d) regulatory alternatives are very low overall for the subset of affected closed and existing landfills. Costs per household in absolute and relative terms are low for the households served by most affected landfills. Similarly, the capital costs of the alternatives under consideration are also low relative to net municipal debt.

8.6.2 Section 111(b) Standards

Table 8-16 in Sec. 8.4.2 indicates that the 25, 100, and 250 Mg stringency levels for the §111(b) Standards affect only 26%, 11%, and 4%, respectively, of all the new landfills in the United States between 1992

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*The landfills having control costs in excess of 10% of local taxes paid by households are the same landfills having relatively high control costs per household. As explained above, the relatively high annualized costs as a percentage of local taxes are attributable to overestimated control costs resulting from overestimated MSW acceptance rates and/or underestimated local taxes as a result of underestimated numbers of households served by these landfills.

*The seven landfills in this category at the 100 Mg stringency level are the result of scaling the estimated capital costs of emissions controls as a percentage of net municipal debt at one landfill in the database. This landfill has an extremely high MSW acceptance rate relative to the number of households it serves. Thus, its high capital costs as a percentage of net municipal debt is probably attributable to overestimated capital costs as a result of an overestimated MSW acceptance rate and/or an underestimate of net municipal debt as a result of an underestimate of the number of municipalities served by this landfill.

TABLE 8-31. NET PRESENT VALUE OF CAPITAL COSTS AS A PERCENTAGE OF NET MUNICIPAL DEBT FOR A SUBSET OF AFFECTED PUBLICLY OWNED CLOSED AND EXISTING LANDFILLS

	Stringency Levels (Mg NMOC/yr)		
	25	100	250
National average capital cost as a percent of net municipal debt (%)	1.9	2.4	1.6
Distribution of affected landfills by capital cost as a percent of net municipal debt (%)			
≤ 1	150 (18)	29 (9)	14 (16)
1 to 5	334 (40)	169 (52)	41 (47)
5 to 25	257 (31)	82 (25)	15 (17)
25 to 100	60 (7)	39 (12)	17 (20)
> 100	36 (4)	7 (2)	0 (0)
Total	837 (100)	326 (100)	87 (100)

Note: Numbers in parentheses are percentages. Net present value of capital cost for publicly owned landfills is calculated using a 4 percent discount rate. Details may not add to totals due to rounding.

and 1997. Since the total number of affected new landfills is relatively small, it is very unlikely that any of the stringency levels will affect more than 20% of the small landfills for the reasons described in Section 8.6.1. We confirmed this tentative conclusion with an analysis of the distribution of affected new landfills by their design capacity relative to the total number of new landfills in the same size categories. Thus, the §111(b) Standards under consideration do not require a Regulatory Flexibility Analysis, because they do not affect a significant number of small entities.

Although a Regulatory Flexibility Analysis is not required for the §111(b) Standards under consideration, we examine the distributional impacts of the various stringency levels for a subset of the affected new landfills (i.e., those landfills for which OSW developed household and governmental data for a landfills rulemaking under Subtitle D of RCRA).* These distributional impacts are:

- population of the service area
- annualized control costs per household
- annualized control costs as a percentage of annual local taxes paid by households
- net present value of capital costs as a percentage of net municipal debt (for publicly owned landfills).

We examined these same distributional impacts for the §111(d) regulatory alternatives in Section 8.6.1.

Table 8-32 presents the population of the service area for the subset of affected new landfills. While a third of the affected new landfills for the 25 Mg stringency level serve 10,000 people or less, none of the affected landfills under the other stringency levels serve such small communities. In general, the 25 Mg stringency level affects smaller communities than the 100 and 250 Mg alternatives. More than two-thirds of

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*As observed for the closed/existing landfills, the affected new landfills for which OSW data are available are generally smaller than the other affected landfills. However, Student-t tests revealed no significant size differences for any of the stringency levels under consideration.

TABLE 8-32. SERVICE AREA POPULATION FOR A SUBSET OF THE AFFECTED NEW LANDFILLS

	Stringency Levels (Mg NMOC/yr)		
	25	100	250
National average service area population (10^3 people)	53.7	93.5	92.5
Distribution of affected landfills by service area population (10^3 people)			
≤ 10	43 (34)	0 (0)	0 (0)
10 to 25	17 (13)	10 (16)	10 (42)
25 to 50	46 (36)	33 (52)	2 (8)
50 to 150	2 (2)	0 (0)	0 (0)
150 to 500	10 (8)	10 (16)	10 (42)
> 500	10 (8)	10 (16)	2 (8)
Total	128 (100)	63 (100)	24 (100)

Note: The numbers in parentheses are percentages. Details may not add to totals due to rounding.

the affected landfills under the 100 Mg stringency level serve communities with 10,000 to 50,000 people.

The national average annualized cost per household for the subset of affected new landfills is below $11 for all three stringency levels (see Table 8-33). As the stringency level decreases, the national average annualized household cost also decreases. Over half the affected landfills under the 100 Mg stringency level have annualized costs per household of $25 or less. However, annualized household costs exceed $50 for 16% of the affected new landfills, ranging as high as $76 per household per year.*

Table 8-34 shows that the national average annualized enterprise cost as a percent of local taxes paid by households is below 1% for the subset of affected new landfills under all three stringency levels. Control costs as a percent of local taxes are under 1% for the households served by almost three-fourths of the affected landfills for the 100 Mg stringency level. Only one-ninth of the affected landfills have control costs as a percent of local taxes paid by households above 10%, with 15% being the maximum.†

The final measure of the distributional impact of the §111(b) Standards under consideration is the net present value of capital costs as a percentage of net municipal debt for a subset of affected, publicly owned, new landfills. Table 8-35 shows that these capital costs are about 2% of net municipal debt as a national average for the affected new landfills. While over four-tenths of the affected new landfills have capital costs under 1% of net municipal debt under the 100 Mg stringency level, the

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*The number of households served by landfills having annual household costs above $25 at the 100 Mg stringency level is very low compared to the amount of MSW going into these landfills. So the relatively high costs for these landfills are a result of overestimated control costs caused by overestimated MSW acceptance rates and/or underestimated numbers of households served by these landfills.

†The seven landfills in this category at the 100 Mg stringency level are the result of scaling the annualized costs as a percentage of local taxes per household at one landfill in the database. This landfill has a very low amount of local taxes per household (i.e., $105). Consequently, its costs-compared-to-taxes percentage is relatively high.

TABLE 8-33. ANNUALIZED ENTERPRISE COSTS PER HOUSEHOLD FOR A SUBSET OF THE AFFECTED NEW LANDFILLS

	Stringency Levels (Mg NMOC/yr)		
	25	100	250
National average annualized costs per household ($/Household)	10.56	8.55	8.37
Distribution of affected landfills by annualized cost per household ($/Household)			
≤ 5	29 (23)	17 (27)	10 (42)
5 to 10	22 (17)	22 (35)	3 (13)
10 to 25	24 (19)	7 (11)	2 (8)
25 to 50	22 (17)	7 (11)	7 (29)
> 50	31 (24)	10 (16)	2 (8)
Total	128 (100)	63 (100)	24 (100)

Note: Numbers in parentheses are percentages. Costs for publicly owned landfills are annualized at 4 percent over the control period. Costs for privately owned landfills are annualized at 8 percent over the life of the landfill. Details may not add to totals due to rounding.

TABLE 8-34. ANNUALIZED ENTERPRISE COST AS A PERCENTAGE OF LOCAL TAXES PAID BY HOUSEHOLDS IN THE SERVICE AREA FOR A SUBSET OF THE AFFECTED NEW LANDFILLS

	Stringency Levels (Mg NMOC/yr)		
	25	100	250
National average annualized enterprise cost as a percent of taxes paid by households (%)	0.8	0.7	0.5
Distribution of affected landfills by average annualized cost as a percent of taxes paid by households (%)			
≤ 1	70 (55)	46 (73)	15 (63)
1 to 2.5	34 (27)	7 (11)	7 (29)
2.5 to 10	10 (8)	3 (5)	2 (8)
> 10	14 (11)	7 (11)	0 (0)
Total	128 (100)	63 (100)	24 (100)

Note: Numbers in parentheses are percentages. Costs for publicly owned landfills are annualized at 4 percent over the control period. Costs for privately owned landfills are annualized at 8 percent over the active life of the landfill. Details may not add to totals due to rounding.

TABLE 8-35. NET PRESENT VALUE OF CAPITAL COST AS A PERCENTAGE OF NET MUNICIPAL DEBT FOR A SUBSET OF AFFECTED PUBLICLY OWNED NEW LANDFILLS

	Stringency Levels (Mg NMOC/yr)		
	25	100	250
National average capital cost as a percent of net municipal debt (%)	2.1	2.2	1.4
Distribution of affected landfills by capital cost as a percent of net municipal debt (%)			
≤ 1	20 (22)	17 (41)	3 (25)
1 to 2.5	31 (34)	7 (17)	0 (0)
2.5 to 10	22 (24)	7 (17)	7 (58)
> 10	17 (19)	10 (24)	2 (17)
Total	90 (100)	41 (100)	12 (100)

Note: Numbers in parentheses are percentages. Net present value of capital cost for publicly owned landfills is calculated using a 4 percent discount rate. Details may not add to totals due to rounding.

capital costs for almost one-quarter of the affected new landfills under this stringency level are more than 10% of net municipal debt.*

In summary, the distributional impacts of the §111(b) regulatory alternatives are very low overall for the subset of affected new landfills. Costs per household in absolute and relative terms are low for the households served by almost all the affected new landfills. Similarly, the capital costs of the regulatory alternatives under consideration are also low relative to net municipal debt.

8.7 DISCOUNT RATE SENSITIVITY ANALYSIS

Section 8.4 analyzes the net present value of social costs for affected landfills calculated using a two-stage discounting procedure. First, we annualized capital costs over the control period using a 10% discount rate. Then, we discounted the sum of annualized capital costs and annual operating costs at 3% to obtain the net present value of social costs. To investigate the sensitivity of capital costs, operating costs, and total costs to changes in the discount rate, we recalculated social costs using a single discount rate applied to both capital and operating costs.

8.7.1 Section 111(d) Guidelines

Table 8-36 contains the net present value of social costs using a 3% discount rate for affected closed and existing landfills for each §111(d) regulatory alternative under consideration. The costs in this table show a significant decrease in capital costs compared to the costs in Table 8-15 (net present value of social costs using two-stage discounting). Operating costs are discounted using 3% in both cases, so there is no difference between the operating costs presented in these tables. Table 8-37 shows the effect of a 10% discount rate on the net present value of social cost. This table shows a further reduction in capital costs as well as a

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*The 10 landfills in this category at the 100 Mg stringency level are the result of scaling the capital costs of two landfills in the database. Both these landfills have extremely high MSW acceptance rates relative to the number of households they serve. So their relatively high capital costs compared to net municipal debt are probably attributable to overestimated capital costs as a result of overestimated MSW acceptance rates and/or an underestimate of net municipal debt as a result of an underestimate of the number of municipalities served by these landfills.

TABLE 8-36. NET PRESENT VALUE OF SOCIAL COSTS FOR AFFECTED CLOSED AND EXISTING LANDFILLS USING A THREE PERCENT DISCOUNT RATE

Net Present Value	Stringency Levels (Mg NMOC/yr) 25	100	250
National social costs (10^6)			
Capital	2,473	1,764	963
Operating	5,213	2,831	1,514
Total	7,686	4,595	2,477
Average total social cost per affected landfill (10^6)	4.08	5.39	6.42
Distribution of affected landfills by net present value of social costs (10^6)			
≤ 0.5	53	29	7
	(3)	(3)	(2)
0.5 to 1.0	131	46	7
	(7)	(5)	(2)
1.0 to 3.0	850	283	119
	(45)	(33)	(31)
3.0 to 5.0	508	242	135
	(27)	(28)	(35)
5.0 to 10.0	265	185	79
	(14)	(22)	(20)
>10.0	77	68	39
	(4)	(8)	(10)
Total	1,884	853	386
	(100)	(100)	(100)

Note: Numbers in parentheses are percentages. Net present value of social costs are computed using a 3 percent discount rate. Details may not add to totals due to rounding.

TABLE 8-37. NET PRESENT VALUE OF SOCIAL COSTS FOR AFFECTED CLOSED AND EXISTING LANDFILLS USING A TEN PERCENT DISCOUNT RATE

Net Present Value	Stringency Levels (Mg NMOC/yr) 25	100	250
National social costs (10^6)			
Capital	1,812	1,318	719
Operating	1,569	906	470
Total	3,381	2,224	1,189
Average total social cost per affected landfill (10^6)	1.79	2.61	3.08
Distribution of affected landfills by net present value of social costs (10^6)			
≤ 0.5	286 (15)	111 (13)	32 (8)
0.5 to 1.0	683 (32)	140 (16)	41 (11)
1.0 to 3.0	783 (42)	433 (51)	232 (60)
3.0 to 5.0	132 (7)	104 (12)	43 (11)
5.0 to 10.0	53 (3)	38 (4)	19 (5)
>10.0	27 (1)	27 (3)	19 (5)
Total	1,884 (100)	853 (100)	386 (100)

Note: Numbers in parentheses are percentages. Net present value of social costs are computed using a 10 percent discount rate. Details may not add to totals due to rounding.

significant reduction in operating costs when compared with the two-stage results. For the 100 Mg stringency level in particular, going from two-stage to single-stage discounting using a 3% discount rate reduces the average cost by 36%; using a 10% discount rate reduces the average cost by 69%.

We estimated annualized social costs by applying an annualization factor to the net present value of total social costs. In all cases we annualized social costs from 1992 to the end of each landfill's control period. Table 8-38 compares costs calculated using two-stage discounting, single-stage discounting at 3%, and single-stage discounting at 10% for affected closed and existing landfills. As expected, two-stage discounting results in higher costs than either of the single-stage calculations. However, annualized costs calculated using a 3% discount rate are lower than annualized costs calculated using a 10% discount rate because of the variable annualization period across affected landfills.

8.7.2 Section 111(b) Standards

Tables 8-39 and 8-40 contain the results of calculating the net present value of social costs for affected new landfills using a 3% and 10% discount rate, respectively. Comparing costs in Table 8-39 with those in Table 8-22 (net present value of social costs using two-stage discounting) shows a decrease in capital costs, but no change in operating costs. Table 8-40 shows a further reduction in capital costs as well as a significant reduction in operating costs when compared with the two-stage results. For the 100 Mg stringency level in particular, going from two-stage to single-stage discounting using 3% reduces the average cost by 37%; using a 10% discount rate reduces the average cost by 83%.

Table 8-41 compares annualized social costs for affected new landfills using different discount rates. As expected, two-stage discounting results in higher costs than the single-stage discounting. Unlike the results for affected closed/existing landfills, the single-stage annualized costs for affected new landfills follow the same pattern as the net present value of costs. That is, annualized costs calculated using a 3% discount rate are higher than those calculated using a 10% discount rate.

TABLE 8-38. TOTAL ANNUALIZED SOCIAL COST FOR AFFECTED CLOSED AND EXISTING LANDFILLS USING VARIOUS DISCOUNT RATES (10^6)

	Stringency Level (Mg NMOC/yr)		
	25	100	250
2-Stage Discounting*	416	297	150
3% Discount Rate**	281	202	99
10% Discount Rate**	358	257	129

* Two-stage discounting involves annualizing each landfill's capital costs at 10% over its control period. Then net present values are computed by discounting annual operating costs and annualized capital costs at 3%. Finally, the net present values are annualized at 3% from 1992 to the end of each landfill's control period and then summed.

** Net present values are annualizd from 1992 to the end of each landfill's control period and then summed.

TABLE 8-39. NET PRESENT VALUE OF SOCIAL COSTS FOR AFFECTED NEW LANDFILLS USING A THREE PERCENT DISCOUNT RATE

Net Present Value	Stringency Levels (Mg NMOC/yr) 25	100	250
National social costs (10^6)			
Capital	299	215	143
Operating	614	348	200
Total	913	564	343
Average total social cost per affected landfill (10^6)	3.7	5.4	8.4
Distribution of affected landfills by net present value of social costs (10^6)			
≤ 0.5	14 (6)	0 (0)	0 (0)
0.5 to 1.0	10 (4)	0 (0)	0 (0)
1.0 to 3.0	131 (53)	46 (44)	7 (17)
3.0 to 5.0	60 (24)	34 (33)	22 (54)
5.0 to 10.0	22 (9)	14 (13)	2 (5)
> 10.0	10 (4)	10 (10)	10 (24)
Total	247 (100)	104 (100)	41 (100)

Note: Numbers in parentheses are percentages. Net present value of social cost is calculated using a 3 percent discount rate. Details may not add to totals due to rounding.

TABLE 8-40. NET PRESENT VALUE OF SOCIAL COSTS FOR AFFECTED NEW LANDFILLS USING A TEN PERCENT DISCOUNT RATE

Net Present Value	Stringency Levels (Mg NMOC/yr) 25	100	250
National social costs (10^6)			
Capital	127	90	58
Operating	112	63	35
Total	239	154	93
Average total social cost per affected landfill (10^6)	1.0	1.5	2.3
Distribution of affected landfills by net present value of social costs (10^6)			
≤ 0.5	109	41	7
	(44)	(39)	(17)
0.5 to 1.0	68	17	7
	(28)	(16)	(17)
1.0 to 3.0	53	36	17
	(21)	(35)	(41)
3.0 to 5.0	10	3	3
	(4)	(3)	(7)
5.0 to 10.0	7	7	7
	(3)	(7)	(17)
> 10.0	0	0	0
	(0)	(0)	(0)
Total	247	104	41
	(100)	(100)	(100)

Note: Numbers in parentheses are percentages. Net present value of social cost is computed using a 10 percent discount rate. Details may not add to totals due to rounding

TABLE 8-41. TOTAL ANNUALIZED SOCIAL COSTS FOR AFFECTED NEW LANDFILLS USING VARIOUS DISCOUNT RATES (10^6)

	Stringency Level (Mg NMOC/yr)		
	25	100	250
2-Stage Discounting*	45.2	30.2	19.0
3% Discount Rate**	29.6	19.2	11.8
10% Discount Rate**	23.9	15.5	9.3

* Two-stage discounting involves annualizing each landfill's capital costs at 10% over its control period. Then net present values are computed by discounting annual operating costs and annualized capital costs at 3%. Finally, the net present values are annualized at 3% from 1992 to the end of each landfill's control period and then summed.

** Net present values are annualizd from 1992 to the end of each landfill's control period and then summed.

8.8 SUMMARY AND CONCLUSIONS

We focused our economic analysis on the flare option for controlling NMOC emissions from closed/existing and new landfills, although we also presented results for a cost-minimizing energy recovery option for the subset of affected landfills having positive energy recovery costs. The flare option assumes that all affected landfills will control NMOC emissions using flares, which overestimates the actual cost of the regulatory alternatives because some landfills will choose a cheaper energy recovery option. As explained in Section 8.3, our energy recovery option underestimates the costs of the regulatory alternatives at some landfills and overestimates compliance costs at other landfills, with the aggregate effect being unknown. Although EPA emissions controls will increase the likelihood that landfills will select an energy recovery option, there is no way to accurately predict which affected closed/existing and new landfills will actually select this option.

As discussed in Section 8.3, two features of the costing model presented in Chapter 7 are noteworthy for the economic analysis. First, the model assumes that landfills that close between 1987 and 1997 are replaced by an identical landfill serving the same area, while recent evidence indicates that the number of U.S. landfills is actually declining. The model also uses relatively high MSW acceptance rates, which is an important parameter in determining NMOC emissions rates and the cost of emissions controls. These features lead to overestimates of the number of affected landfills, compliance costs, and emissions reductions.

In summary, the actual economic impacts of the §111(d) and 111(b) regulatory alternatives under consideration are probably less than the economic impacts presented in this chapter. Nevertheless, our analysis of these regulatory alternatives leads to several specific conclusions:

- The regulatory alternatives will affect only a small fraction of the closed/existing and new landfills (generally less than 15%), and most of the affected landfills are relatively large.

- The number of affected closed, private landfills, which have no way of generating revenues to cover compliance costs, is small under the flare option and even smaller under the energy recovery option.

- Most control periods are relatively long under the various stringency levels and control options, with most of the control period coming after the closure of affected landfills.

- The national net present value of enterprise costs decreases substantially as the stringency level decreases under both control options for affected closed/existing and new landfills, but the average enterprise cost rises as the stringency level decreases.

- The national annualized enterprise control cost per Mg of MSW is below $1 per Mg for stringency levels under the flare option for affected existing and new landfills and for stringency levels under the energy recovery option for affected new landfills. National annualized enterprise control costs per Mg of MSW range between $1.43/Mg and $2.66/Mg for affected existing landfills under the energy recovery option.

- The costs of the regulatory alternatives are very low for most households--the majority of affected existing landfills have compliance costs under $15 per household per year and the majority of affected new landfills have compliance costs under $10 per household per year.

- While the national cost-effectiveness of almost all the stringency levels under both the flare and energy recovery options is less than $1000 per Mg of NMOC emissions reduction, cost effectiveness varies greatly among affected landfills--much more than is typical for EPA stationary-source regulations.

- The regulatory alternatives under consideration for closed/existing and new landfills will not affect a substantial number of small entities, so a Regulatory Flexibility Analysis is not required for either the §111(d) or 111(b) rulemakings.

- The social costs of the regulatory alternatives for affected closed/existing and new landfills are very sensitive to the discount rate, because of the long control periods under stringency levels for both the flare and energy recovery control options.

In general, the economic impacts of the §111(d) and 111(b) regulatory alternatives on households and municipalities are too small to significantly influence the choice among these alternatives. Privately owned landfills that are already closed and must install emissions controls may be significantly impacted by the regulatory alternatives, because they have no way of recovering their compliance costs. However, there are very few closed, privately owned landfills that are affected under any of the

regulatory alternatives. The control costs of the regulatory alternatives at affected landfills will probably not lead to a significant shift in MSW flows from landfills to municipal waste combustors. Finally, all of the regulatory alternatives will stimulate the adoption of energy recovery technologies at affected landfills.

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9. GUIDANCE FOR IMPLEMENTING THE EMISSION GUIDELINES AND COMPLIANCE SCHEDULE

This chapter, in concert with the entire background information document, has been prepared in accordance with regulations established under Section 111(d) of the Clean Air Act. Under the regulations contained in Subpart B of 40 CFR 60, EPA has established procedures whereby States submit plans to control existing sources of "designated pollutants". Designated pollutants are pollutants which are not included on a list published under Section 108(a) (National Ambient Air Quality Standards) or 112(b)(1)(A) (Hazardous Air Pollutants), but to which a standard of performance for new sources applies under Section 111(b). Under Section 111(d), emission standards are to be adopted by the States and submitted to EPA for approval. The standards would limit the emissions of designated pollutants from existing facilities which, if new, would be subject to the standards of performance for new stationary sources. Such facilities are called designated facilities. The purpose of this chapter is to provide guidance in implementing the emission guidelines and compliance schedules for existing municipal solid waste landfills, and to provide information upon which States may base their plans. The guidance provided in this chapter also applies to new municipal solid waste landfills.

After public review and comment on the draft emission guidelines, a final guideline will be published, and the emission guideline and compliance schedule will be promulgated under Subpart C of 40 CFR 60. The States will then have nine months to develop and submit plans for control of the designated pollutant (municipal landfill gas emissions) from designated facilities. Within four months after the date for submission of such plans, the Administrator will approve or disapprove each plan (or portions thereof). If a State plan (or portion thereof) is disapproved, the Administrator will promulgate a plan (or portion thereof) within six months after the date for plan submission. These and related provisions of Subpart B are basically patterned after Section 110 of the Act and 40 CFR 51 (concerning the adoption and submittal of State implementation plans under Section 110).

As discussed in the preamble to Subpart B (40 FR 5340, November 17, 1975), a distinction is drawn between designated pollutants which may cause or contribute to endangerment of public health (referred to as "health-related pollutants") and those for which adverse effects on public health have not been demonstrated (referred to as "welfare-related pollutants"). For health-related pollutants, emission standards and compliance times in State plans must be at least as stringent as the corresponding emission guidelines and compliance times in EPA's guideline document, but 40 CFR 24.(g) does allow States to adopt and enforce emissions standards and compliance times which are more stringent than those provided in the published guidelines. In addition, as provided in Subpart B, States may apply less stringent requirements for particular designated facilities or classes of facilities, on a case-by-case basis, when economic factors or physical limitations make such less stringent control more reasonable. Such justification may include unreasonable control costs resulting from plant age, location, process design, or the physical impossibility of installing the specified control system. States may also relax compliance time if sufficient justification is provided. Justification for such a relaxation may include unusual time delays caused by unavailability of labor, climatological factors, scarcity of strategic materials, and large work backlogs for vendors or contractors.

For reasons discussed at length in Chapter 2 of this background information document, the Administrator has determined that air emissions from municipal solid waste landfills are health-related pollutants. Briefly, this determination is based on four specific health and welfare effects attributable to these emissions: (1) the adverse health and welfare effects resulting from nonmethane organic emissions, (2) the contribution to global warming of methane emissions, (3) explosion hazard, and (4) odor nuisance. Therefore, the States must develop regulations to control these emissions that are at least as stringent as the final guidelines.

The guidance document mandated under Subpart B must provide specific information to assist States in the development of a plan under Section 111(d). Much of this information is nearly identical for both

new and existing landfills, and has already been provided in this background information document as listed below:

	BID Chapter(s)
Health and welfare effects of air emissions of MSW Landfills	Chapter 2
Landfill gas collection and control techniques	Chapter 4
Control technology efficiency and environmental effects	Chapter 6
National emission reduction potential of guideline	Chapter 6

Rather than duplicate the information which is already provided in this BID, this chapter will focus on the following:

- Time necessary for normal design, installation, and start-up of identified collection and control systems.
- An emission guideline reflective of Best Demonstrated Technology (BDT), and a compliance guideline.

The guidance presented in this section applies to all existing municipal solid waste (MSW) landfills that accepted refuse at any time between November 8, 1987 and the date of proposal of the New Source Performance Standards (NSPS) for MSW landfills. Existing landfills that have capacity available and are not closing prior to accepting any additional refuse are also affected. Landfills which commence construction, or in the absence of construction received refuse, on or after the date of proposal (the NSPS) are defined as new landfills and are subject to the NSPS. The requirements for new landfills are identical to those for existing landfills.

Only a portion of the existing landfills subject to the emission guidelines are required to install air emission control systems. This is the subset of existing municipal solid waste landfills with the greatest potential for adversely impacting public health and welfare. However, many of the landfills included under this definition of designated facility may not pose a significant threat to public health and welfare. The public

health and welfare threat posed by individual municipal solid waste landfills varies widely and more specific guidance on if and when air emission control systems are required at a specific landfill is provided in Section 9.1.

For those facilities required to install landfill gas collection and control systems, specific guidelines for the design and operation of these systems are provided in Sections 9.2, 9.3, and 9.4. The guidelines are separated into two distinctive components: guidelines for effective collection of the municipal landfill gas; and control of the collected landfill gas. Section 9.2 provides guidelines on the design of an effective gas collection system. Section 9.3 provides guidelines on effective operation of the gas collection system. Section 9.4 provides design and operating guidelines for the air emission control device.

Finally, the schedule for compliance with these emission guidelines is presented in Section 9.5. A schedule for compliance is provided for both initial installation of the collection/control system and continued expansion of the collection/control system, as new refuse is placed in active portions of the landfill.

9.1 DETERMINATION OF CONTROL REQUIREMENT

The owner or operator of a designated MSW landfill with a maximum design capacity less than 100,000 Mg refuse must submit a report to the State agency documenting the landfill size. Documentation should include a map or plot of the landfill which provides the size and location of the landfill and identifies all areas where refuse may be landfilled as permitted by the state or county. Documentation should also include the maximum design capacity as specified in the State or county or RCRA permit. If the design capacity has not been specified, then the capacity should be estimated and a copy of the estimation method submitted for review. Upon the State's verification that the maximum design capacity of the landfill is less than 100,000 Mg, the landfill owner/operator is not required to perform further testing reporting, or to install controls. If the design capacity is increased by the addition of new areas, by an increase in the depth of

refuse deposition, by greater compaction, or any other means, an amended design capacity report must be submitted. If the revised capacity exceeds 100,000 Mg, the landfill would then be subject to the additional provision of the guideline.

The owner or operator of a designated MSW landfill with a maximum design capacity greater than 100,000 Mg refuse is required to periodically determine the nonmethane organic compound (NMOC) emission rate from his/her landfill each year, from the effective date of an approved State plan for implementing the emission and compliance guidelines until closure of the landfill. This includes landfills with an existing collection/control system in place. A procedure for determining periodic NMOC emission rate is provided in Section 9.1.1 below. The determined NMOC emission rate is to be reported to the State each year along with supporting data and calculations.

If the NMOC emission rate is determined to be greater than or equal to 150 Mg of NMOC per year, then the landfill owner is required to install a collection system which effectively captures the generated gas and conveys this collected gas to a control system capable of achieving at least a 98 percent reduction in NMOC or a 20 ppmv outlet concentration (dry basis) at 3 percent oxygen. A recovery system can be used to process the landfill gas for subsequent sale, but all atmospheric vents from the recovery system are required to be routed to a control system capable of achieving an overall 98 percent reduction in NMOC or 20 ppmv outlet at 3 percent oxygen. Specific design and operating requirements for the collection and control systems are provided in Section 9.2, 9.3, and 9.4.

At landfills with active collection systems in place, the existing collection system can be used to determine the NMOC mass emission rate only if the system is operating according to the guidelines provided in this chapter. Landfills with passive collection systems in place must have synthetic liners on the bottom, sides, and top of the landfill, as well as, meet the operating guidelines in Section 9.3. Use of existing collection equipment to determine the NMOC mass emission rate is discussed separately in Section 9.1.2.

The owner of a regulated landfill is required to operate the collection and control system, in accordance with the operating guidelines, for a minimum of 15 years, until the landfill is no longer accepting waste and until emissions from the landfill are determined to be less than 150 Mg/year. The procedure for determining when control is no longer required is outlined in Section 9.1.3.

9.1.1 NMOC Emission Rate Determination

The NMOC emission rate is to be determined using the tiered approach as illustrated in Figure 9-1. In the first tier (illustrated in Figure 9-2), the landfill owner or operator is to estimate the NMOC emission rate using the following equation, assuming the acceptance rate is constant from year to year:

$$M_{NMOC} = 2L_o R (e^{-kc} - e^{-kt}) (C_{NMOC})(3.595 \times 10^{-9})$$

where,

M_{NMOC} = mass emission rate of NMOC, Mg/yr
L_o = refuse methane generation potential, m^3/Mg refuse
R = average annual acceptance rate, Mg/yr
k = methane generation rate constant, 1/yr
c = years since closure (c = 0 for active and/or new landfills)
t = age of landfill, yrs
C_{NMOC} = concentration of NMOC, ppmv as hexane
3.595×10^{-9} = conversion factor

The average acceptance rate, R, can be determined by dividing the refuse in place by the age of the landfill. This method for determining the emission rate should only be used for landfills with little or no knowledge of the actual year-by-year refuse acceptance rate. If refuse acceptance rate information is available, the landfill owner should determine the methane generation rate for each yearly submass of refuse and total the results to obtain an accurate overall landfill emission rate. The following equation can be used for the submass approach:

$$Q_i = 2 k L_o M_i (e^{-kt_i}) (C_{NMOC}) (3.595 \times 10^{-9})$$

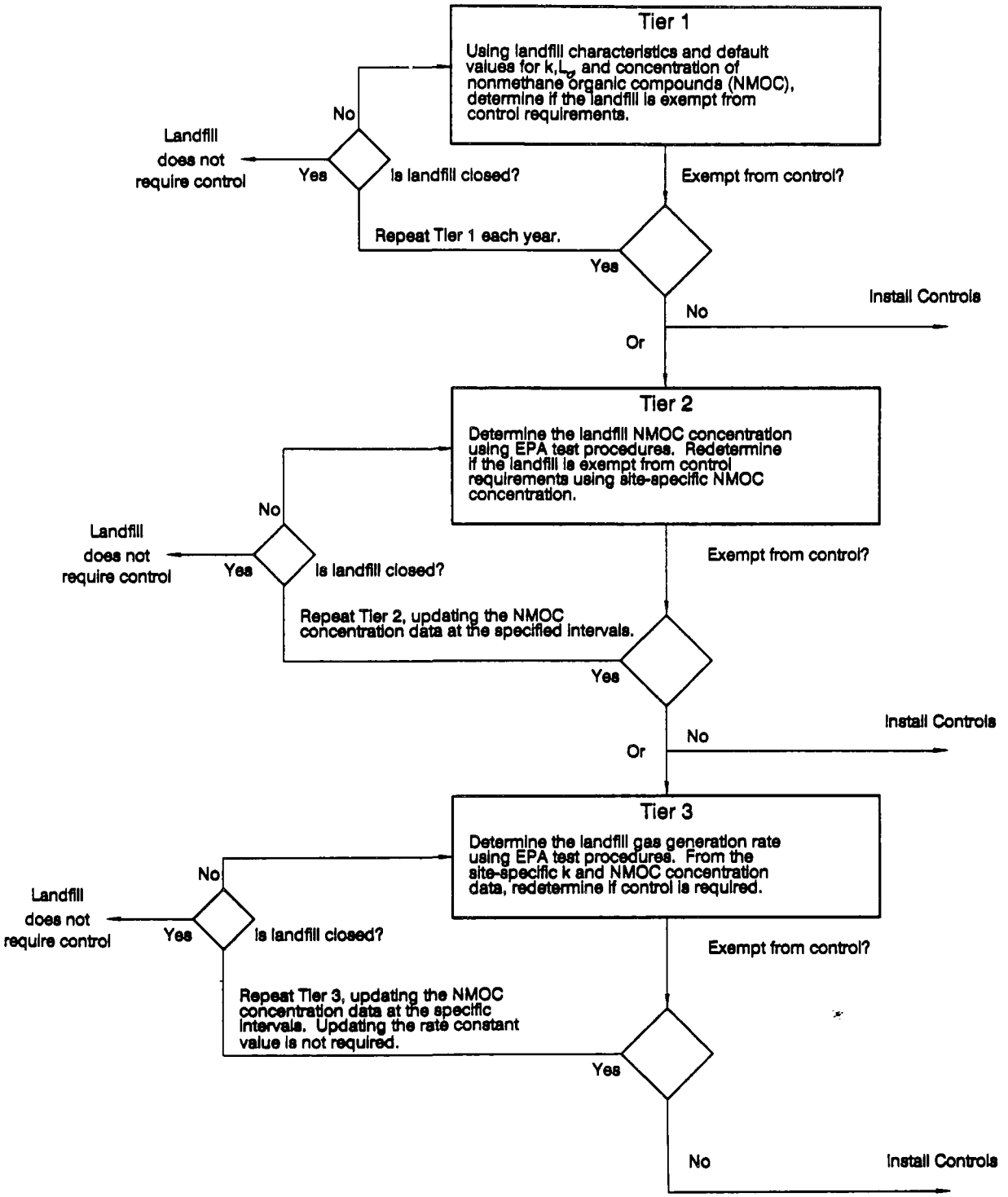

Figure 9-1. Overall Three-Tiered Approach for Determination of Control Requirements

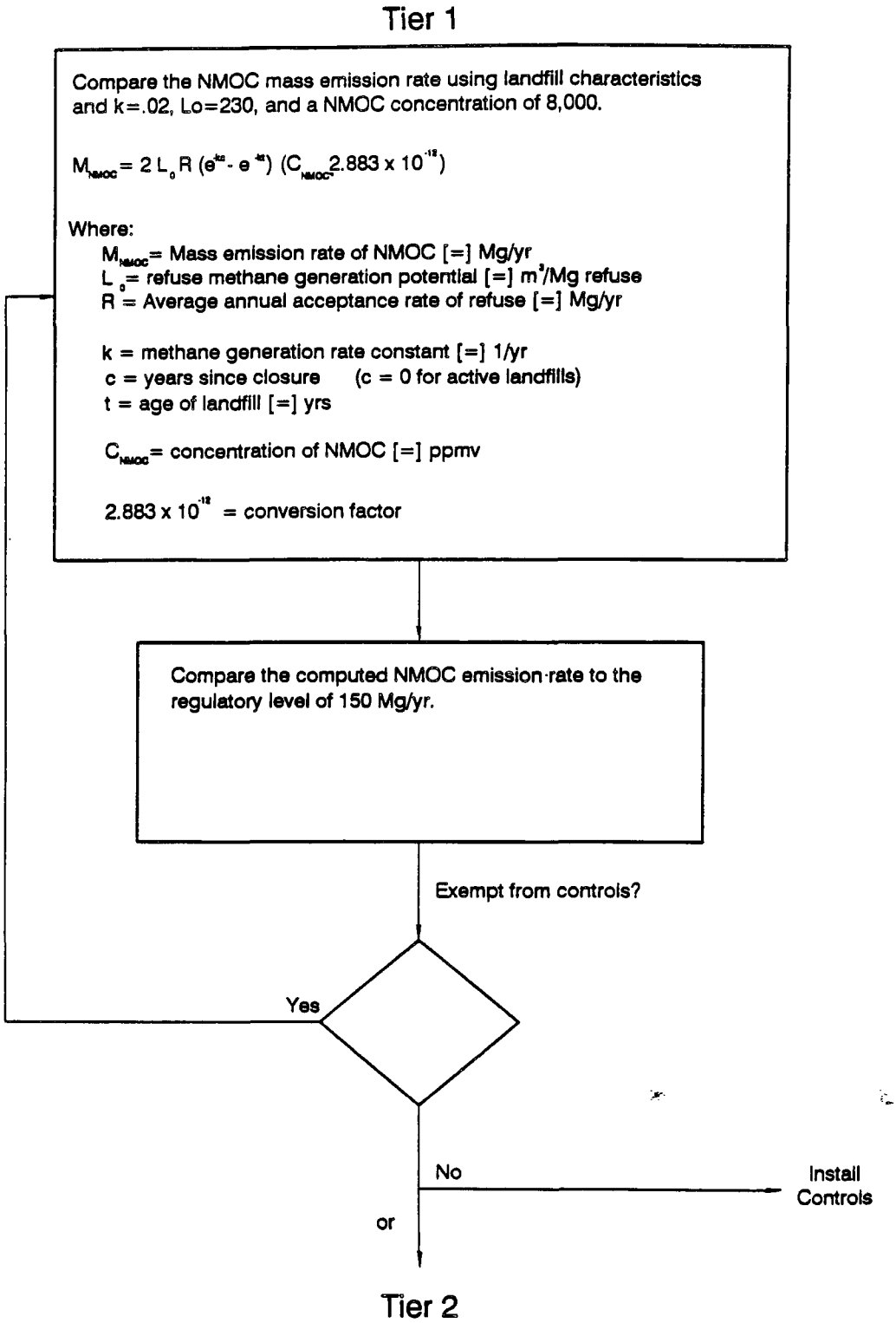

Figure 9-2. Example of Tier 1 Using NMOC Emission Rate Cutoff as the Regulatory Option

where:

Q_i = NMOC emission rate from the ith section, Mg/yr

k = landfill gas generation constant, 1/yr

L_o = methane generation potential, m^3/Mg

M_i = mass of the ith section, Mg

t_i = age of the ith section, yrs

C_{NMOC} = concentration of NMOC, ppmv

3.595×10^{-9} = conversion factor

Regardless of which method is chosen, the nondegradable refuse, such as demolition refuse, should be subtracted from the mass or acceptance rate to avoid overestimating the landfill emission rate. A combination of the two methods may be used if acceptance rate information, such as gate receipts, is only available for a limited time period.

Landfill gas flowrate and/or composition data obtained within 5 years prior to the initial Tier 1 evaluation may be used to determine site-specific values for k and C_{NMOC} provided that the methods used to obtain the data are comparable to EPA Method 2E for flowrate determination and Method 25C for NMOC concentration analysis. The value for k must be computed as outlined in Section 5 of Method 2E regardless of the method used to obtain the raw data. Sufficient documentation of the methods used to obtain these data must be submitted for the State to review. Documentation should include detailed test procedures, test log or data sheets, and any accompanying calculations. In the absence of site-specific data, the values to be used for k, L_o, and NMOC concentration are .02/yr, 230 m^3/Mg, and 8,000 ppmv, respectively. If the calculated NMOC emission rate is greater than 150 Mg/yr, then the landfill owner must either install controls or determine a site-specific NMOC concentration to use in the equation above. If the landfill owner chooses to determine the NMOC concentration, then the steps of Tier 2, illustrated in Figure 9-3, are to be followed. If the NMOC emission rate determined from Tier 2 is greater than 150 Mg/yr, then the landfill owner must either install controls or determine a site-specific gas generation rate constant, k. If the owner chooses to determine k, then the steps of the third tier, illustrated in Figure 9-4, are to be followed. If

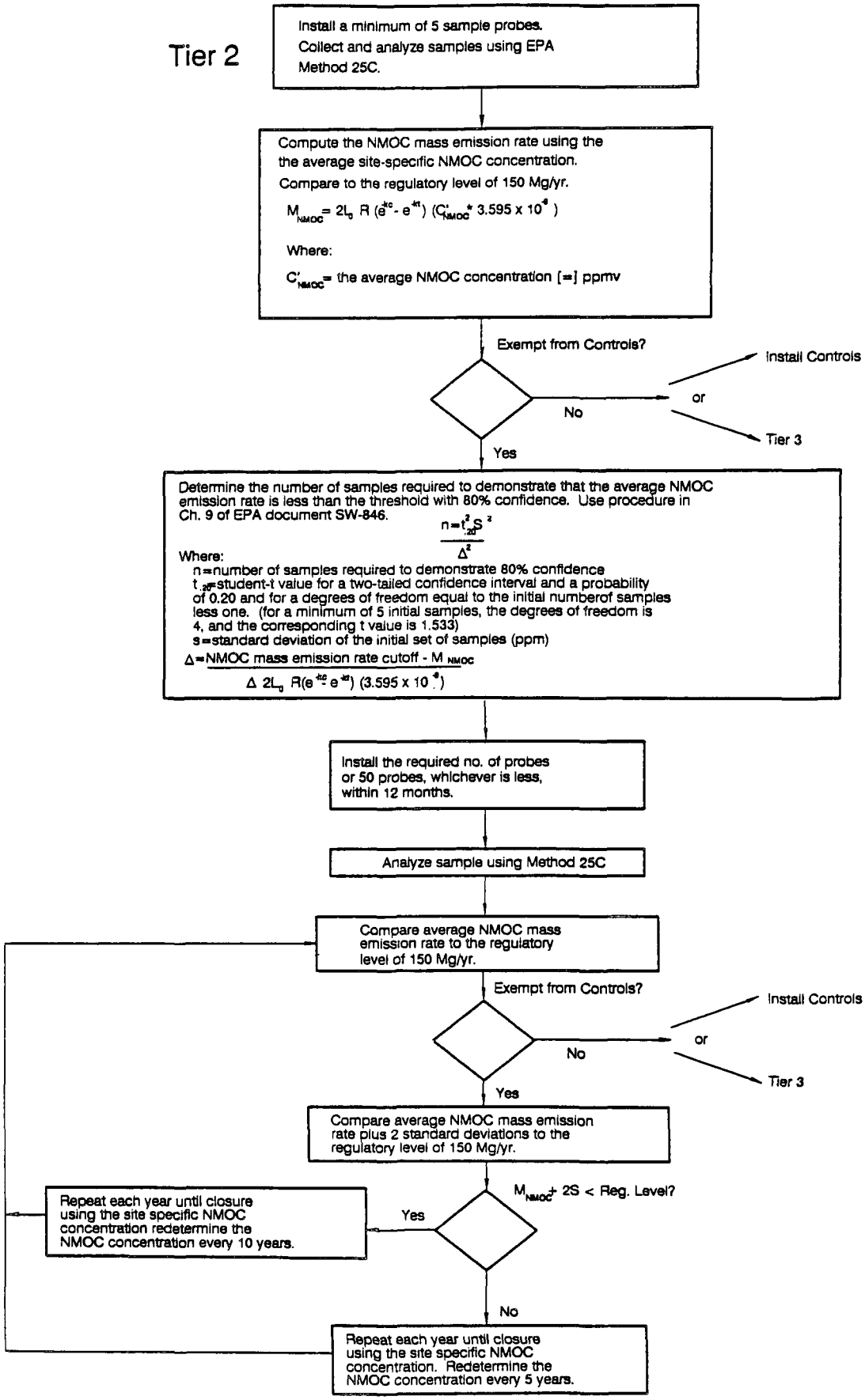

Figure 9-3. Example of Tier 2 Using NMOC Emission Rate Cutoff as the Regulatory Option

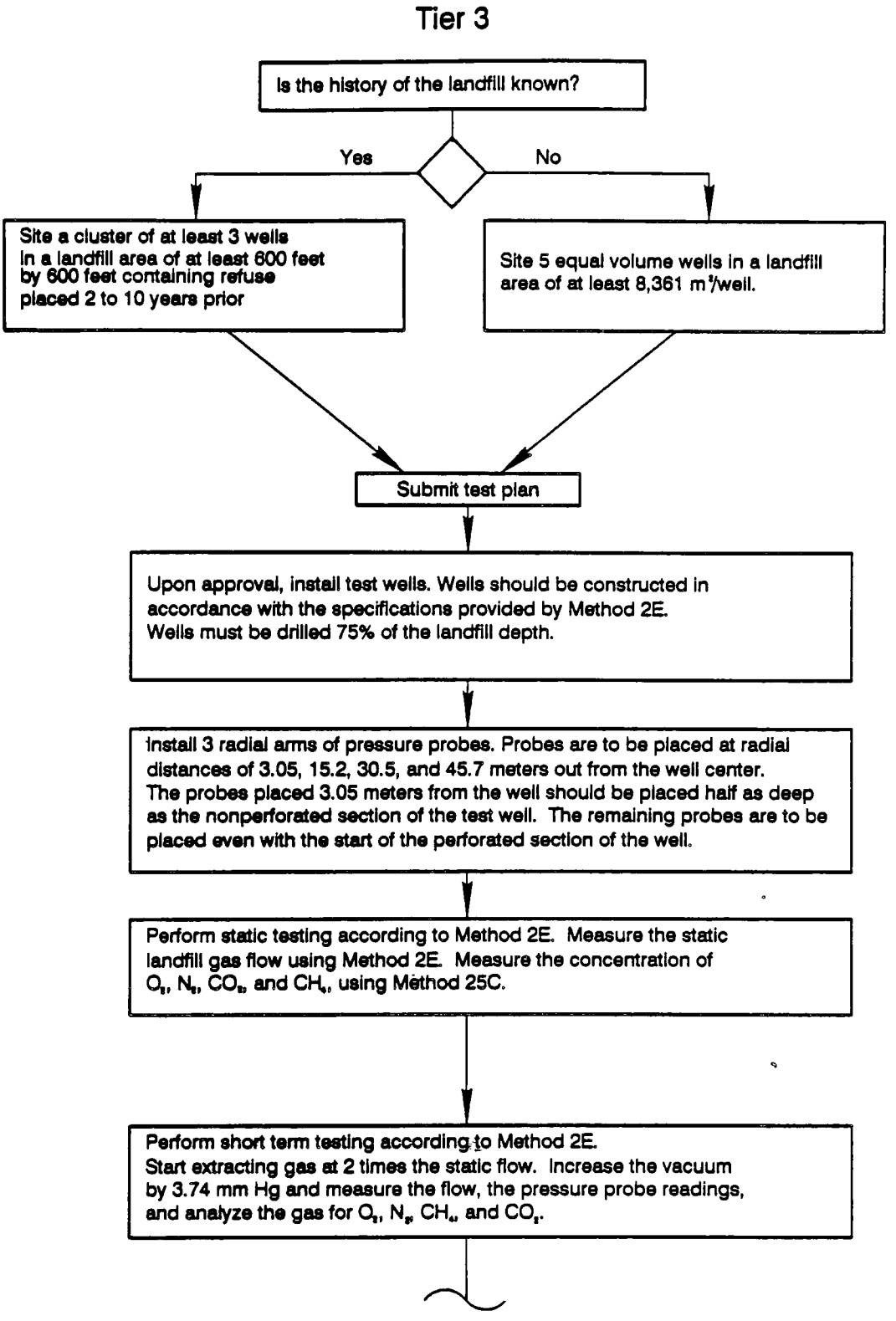

Figure 9-4. Example of Tier 3 Using NMOC Emission Rate Cutoff as the Regulatory Option

When 1% air is detected in the landfill gas or the inner, shallow pressure probe readings show a negative pressure decrease the blower vacuum by 3.74 mm Hg.

↓

Measure the flow, gas composition, and pressure probes daily. Adjust vacuum to maintian steady state conditions.

↓

After achieving steady state for 24 hours determine the radius of influence. The radius of influence is the distance of the deep pressure probe that shows zero differential (i.e. P = P Landfill - P vacuum = 0)

↓

Perform long term multiple-well extraction testing according to Method 2E. extracting the gas at the steady state rate identified in the short term test. Collect and analyze the landfill gas.

History Known →

Calculate CH_4 generation rate constant, k, by trial and error.

$$ke^{kt} = \frac{Q_{test}}{2L_0 M_{test}}$$

Where:

k = CH_4 generation rate constant, 1/yr
Q_{test} = Flowrate for volume tested, m^3/yr
M_{test} = Mass refuse in volume tested, Mg
t = age of volume tested, years
L = refuse methane generation potential [=] m^3/Mg

Calculate total landfill gas flowrate

$$\text{Total } Q_{test} = 2 L_0 R (e^{kt} - e^{-kt})$$

Where:

Q = total flowrate of LFG, m^3/yr
t = age of total landfill yrs

History Not Known →

Calculate total landfill flowrate.

$$Q_{test} = Q_{test} * \frac{\text{Volume of landfill}}{\text{Volume of Test}}$$

the NMOC emission rate determined in Tier 3 is greater than 150 Mg/yr, then controls must be installed in accordance with the compliance schedule provided in Section 9.5.

In determining the NMOC emission rate, the entire municipal solid waste landfill is considered rather than any subdivision of the landfill, such as an individual cell. The entire landfill is defined as the contiguous landfill property designated for solid waste disposal irrespective of subdividing access roads. This includes closed portions of the landfill (no longer accepting refuse), as well as active portions. Additionally, multiple ownership does not affect the definition of a municipal solid waste landfill.

9.1.2 Landfills with a Collection/Control System In Place Prior to Regulation

An owner of a landfill with an existing collection/control system in place has the option of using the tiered approach or using the existing equipment to determine the NMOC mass emission rate for comparison against the standard. The landfill owner may use existing landfill gas collection equipment to determine the NMOC mass emission rate, only if the collection system meets the operating guidelines in Section 9.3. That is, the landfill owner must be able to show that there is not excessive air infiltration and that there is not a positive pressure at each well head. An excessive influx of air may result in an overestimation of the landfill gas flowrate. A positive pressure reading at the well head with a fully open valve means additional wells are required. The landfill owner must also be able to document that the collection system is effectively collecting landfill gas from all gas producing areas of the landfill.

The NMOC mass emission rate can be determined by measuring the total landfill gas flowrate and by determining the NMOC concentration of the gas. The flowrate measurement should be taken at the common header pipe that leads to the control device using an orifice meter as described in Method 2E. The NMOC concentration can be determined by collecting and analyzing a landfill gas sample from the common header pipe using Method 25C. The average NMOC concentration of at least three gas samples

should be used. The following equation can be used to determine the annual NMOC mass emission rate:

$$M_{NMOC} = 1.89 \times 10^{-3} Q_{LFG} C_{NMOC}$$

where:

M_{NMOC} = mass emission rate of NMOC, Mg/yr
Q_{LFG} = flowrate of landfill gas, m^3/min
C_{NMOC} = NMOC concentration, ppmv

If the resulting NMOC mass emission rate is greater than 150 Mg NMOC/yr, then the landfill should continue to operate the collection/control system according to the guidelines outlined in Section 9.3. It is not mandatory that existing collection system meet all of the design specifications included in 9.2, if the collection system meets the operating guidelines provided in Section 9.3. If the NMOC emission rate is less than 150 Mg/yr, then the landfill is exempt from control for that year only. The NMOC mass emission rate should be determined periodically until the landfill closes, and if the NMOC emission rate exceeds 150 Mg/yr at any time, controls should be operated until the requirements of 9.1.3 are met.

9.1.3 Guidelines for Discontinuing Control

Control of landfill air emissions is no longer required when it meets all of the following criteria:

- Controls have been in place and operated for at least 15 years;
- The landfill is no longer accepting waste; and
- Emissions from the landfill are less than 150 Mg/yr.

The annual NMOC mass emission rate must be less than 150 Mg/yr for three consecutive testing periods, between 90 and 180 days apart, in order to meet the emission criteria above.

The emission rate is to be determined by measuring the total landfill gas flowrate and by determining the NMOC concentration of the gas. The flowrate measurement should be taken at the common header pipe that leads to the control device using an orifice meter as described in Method 2E. The NMOC concentration should be determined by collecting and analyzing a gas

sample from the common header pipe using Method 25c. The following equation should be used to determine the annual NMOC mass emission rate for each set of flow and NMOC concentration measurements.

$$M_{NMOC} = 1.89 \times 10^{-3} Q_{LFG} C_{NMOC}$$

where:

M_{NMOC} = mass emission rate of NMOC, Mg/yr
Q_{LFG} = flowrate of landfill gas, m^3/min
C_{NMOC} = NMOC concentration, ppmv

Again, the determined NMOC mass emission rate should be less than 150 Mg/yr for three consecutive quarters before operation of the control system is discontinued.

9.2 DESIGN GUIDELINES FOR GAS COLLECTION SYSTEMS

Landfill gas collection systems can be categorized into two basic types: active collection systems and passive collection systems. Active collection systems employ mechanical blowers or compressors to provide a pressure gradient in order to extract the landfill gas. The systems can be further categorized into two types: vertical well systems and horizontal trench systems. Passive systems rely on the natural pressure gradient (i.e., internal landfill pressure created due to landfill gas generation) or concentration gradient to convey the landfill gas to the atmosphere or to a control system.

The Agency has evaluated the effectiveness of both active and passive collection systems and has concluded that well designed active collection systems are the most effective means of collecting landfill gas.[1] The Agency also found that well designed passive collection systems can approximate the efficiency of an active system when used in conjunction with synthetic liners and caps. Generally, passive collection systems have much lower collection efficiency than active collection systems since they rely on natural pressure gradient (i.e., internal landfill pressure created due to landfill gas generation) or concentration gradient rather than the pressure gradient induced by a blower or compressor. However, the Agency's study revealed that passive collection systems can be nearly equivalent, if the landfill design includes synthetic liners on the top, bottom, and sides

of the landfill. Landfills with highly impermeable containment such as canyons or quarries may also be well-suited for passive systems, however, these should be evaluated on a case-by-case basis taking into account fissures and cracks that may exist in the containment.

Selection of a collection system type often depends on the landfill characteristics and landfill operating practices. For example, if a landfill employs a layer-by-layer landfilling method (as compared to cell-by-cell methods), an active horizontal trench collection system may be preferred over an active vertical well collection system due to the ease of collection system installation. However, if the water table extends into the refuse, horizontal trench systems have a tendency to flood, thus decreasing the collection efficiency. Applications, advantages, and disadvantages of different collection systems are summarized in Table 9-1.

For landfills required to install collection and control systems, the owner of the landfill is first required to develop the collection system design. The design must be based on the specifications for an active vertical collection system provided in Section 60.758 of the NSPS. Alternatively, an owner or operator who wishes to use a collection system not based on those specifications must submit a plan to the State Agency for review. Alternative designs would still need to satisfy the four criteria of an effective collection system provided below, and the plans submitted for review must address each of the four criteria. Provisions for expanding the system as waste accumulates must be indicated in the plan. This plan should include the type of collection system (active or passive), an estimate of the maximum expected gas collection rate, a plot plan of the entire landfill with proposed well placements and estimated radii of influence, and specifications for gas moving equipment. If a passive system is proposed, containment specifications and the estimated collection/control system pressure drop should also be provided. This plan is to be reviewed by the State and, upon approval of the plan, the collection system is to be installed in accordance with the compliance schedule provided in Section 9.5.

The landfill gas collection system must be designed to provide effective collection of the landfill gas. In order for the landfill gas

TABLE 9-1 COMPARISON OF VARIOUS COLLECTION SYSTEMS

Collection system type	Preferred applications	Advantages	Disadvantages
Active Collection Systems			
Vertical Wells	Landfills employing cell-by-cell landfilling methods	Cheaper or equivalent in costs when compared to horizontal trench systems	Difficult to install and operate on the active face of the landfill (may have to replace wells destroyed by heavy operative equipment)
Horizontal Trench	Landfills employing layer-by-layer landfilling methods Landfills with natural depressions such as canyon	Easy to install since drilling is not required Convenient to install and operate on the active face of the landfill	The bottom trench layer has higher tendency to collapse and difficult to repair once it collapses Has tendency to flood easily if water table is high Difficult to maintain uniform vacuum along the length (or width) of the landfill
Passive Collection Systems			
	Landfills with good containment (side liners and cap) Landfills with only gas migration problems	Cheaper to install and maintain if only a few wells are required	Collection efficiency is generally much lower than active collection systems Costs is generally higher than active systems when designed for the same collection efficiency

collection system to be considered effective, it must: (1) provide collection of landfill gas from all gas generating areas within the landfill; (2) provide well spacing adequate to collect landfill gas from all areas of the landfill without overdraw of air into the landfill; (3) provide a gas moving system capable of handling the maximum expected gas flow; and (4) include monitoring and adjustment provisions to facilitate effective operation. Additionally, the gas collection wells are to be constructed in conformance with certain specifications.

The first requirement, collection of landfill gas from all gas producing areas, is common to all collection system types. The gas collection system must be designed to provide gas collection from all gas producing areas of the landfill which contain refuse that is at least two years old. Areas known to contain asbestos should not be included in the collection system design. The collection system should also be designed to extend into each new area of the landfill within two years of the initial placement of refuse in that area. For shallow areas, extraction wells can be installed and vertically extended as more refuse is added. Since this type of installation may make filling that portion of the landfill difficult, it is recommended that the landfill owner/operator manage the filling pattern to avoid shallow sections that meet the age criteria.

Certain landfills will contain sections of refuse that do not produce a significant amount of landfill gas, either due to the age of the refuse or the type of refuse. These "nondegradable" sections may be excluded from control if the landfill owner or operator can show that emissions from the all such sections contribute less than one percent to the total amount of emissions from the landfill. Emissions from a given section may be computed using the following equation:

$$Q_i = 2\ k\ L_o\ M_i\ (e^{-kt_i})\ (C_{NMOC})\ (3.595 \times 10^{-9})$$

where:

Q_i = NMOC emission rate from the ith section, Mg/yr
k = landfill gas generation constant, 1/yr
L_o = methane generation potential, m^3/Mg
M_i = mass of the degradable refuse in the ith section, Mg

t_i = age of the refuse in the ith section, yrs

C_{NMOC} = concentration of NMOC, ppmv

3.595×10^{-9} = conversion factor

The values for k, L_o, and C_{NMOC} used in the tiered procedure should be used if a specific k and C_{NMOC} for the given section has not been determined through field testing. The mass of the nondegradable refuse contained within the given section may be subtracted from the total mass of the section when estimating emissions. The landfill owner or operator should provide records showing the amount and type of refuse claimed as nondegradable and the location of such refuse within the landfill. If more than one section is proposed for exclusion from control, an emissions estimate should be made for each section. The sum of the emissions from all the potentially excluded sections must be less than one percent of the total landfill emissions to qualify for exemption.

The remaining requirements of an effective collection system, adequate well spacing, flow capacity, and well construction are somewhat specific to the type of collection system selected. These requirements are addressed in the following sections specific to each collection system type.

9.2.1 Design Guidelines for Active Vertical Collection Systems

Four design features of the proposed vertical collection system must be evaluated by the owner or operator and by the State reviewer when a collection system design plan is submitted for review to ensure that an effective collection system is installed. These are the proposed well spacing, the proposed well construction, provisions for well monitoring and adjustment, and capacity of the gas mover system. Each of these design features are addressed below.

9.2.1.1 Vertical Well Spacing. The desired method for determining effective well spacing at a specific landfill is the use of field measurement data. EPA Method 2E, prescribed in Tier 3 of the NMOC emission rate determination, can be used to determine the average stabilized radius of influence for both perimeter wells and interior wells. If such a determination has been made using EPA Method 2E, the determined radii of influence are to be used in setting the well spacing. Wells placed along the perimeter of the landfill (but, still in the refuse) are to be placed no

more than the perimeter radius of influence from the perimeter and no more than two times the perimeter radius of influence apart. As illustrated in Figure 9-5, a helpful technique is to site the location of each well and draw a circle with radius equal to the radius of influence (perimeter radius of influence for perimeter wells and interior radius of influence for interior wells). Once the perimeter wells are sited on the landfill plot plan, the interior wells are to be sited at no more than two times the interior radius of influence in an orientation such that essentially all areas of the landfill are covered by the radii of influence. Figure 9-5 provides an illustrative demonstration of this concept.

In situations where the landfill owner chooses not to perform EPA Method 2E, the well spacing must be determined based on theoretical concepts. In order to evaluate the proposed well spacing for these situations, it is important to understand the relationship between applied vacuum (well vacuum) and air infiltration. It is advantageous to apply higher vacuum in order to maximize the radius influence and minimize the number of wells required. But, higher vacuum leads to increased air infiltration. Consequently, excessive air infiltration (greater than one to two percent air) kills the methanogens which produce the landfill gas, supports aerobic decomposition of the refuse, and can potentially lead to a landfill fire.

In the absence of field measurement data, reasonableness of the proposed well vacuum must first be reviewed. The maximum vacuum that can be applied at the well, without excessive air infiltration, is restricted primarily by three landfill characteristics: the landfill depth, gas permeability of the cover or cap material, and the cover thickness. Assuming a 2 ft final cover as required under RCRA, the theoretical vacuum that can be applied without excessive air infiltration is presented in Figure 9-6 for three cover materials. As illustrated in the figure, the maximum vacuum is greatly a function of landfill depth. The maximum vacuum that can be applied is also dependent on the landfill gas generation rate. However, since this can only be determined for a specific site through field measurement, the figure is based on the Scholl-Canyon model with a rate constant (k) of .02 years^{-1} and an ultimate gas generation constant (L_0) of

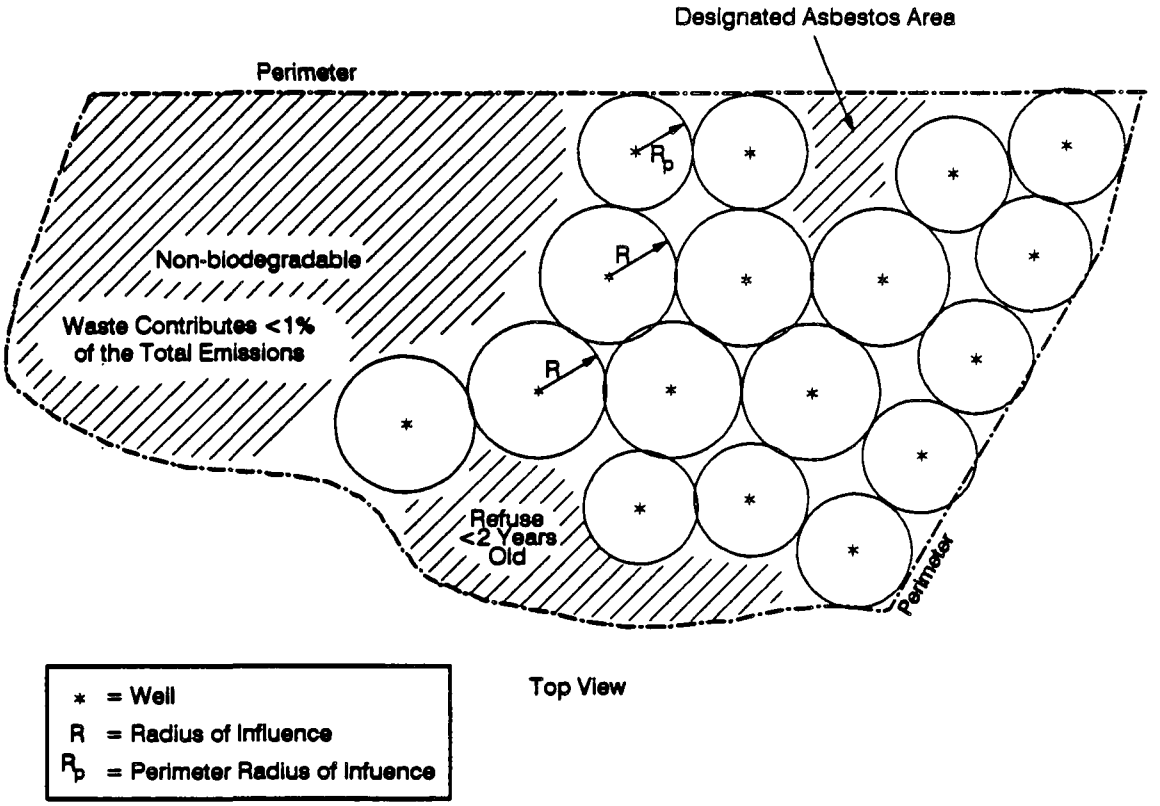

Figure 9-5. Technique for siting wells.

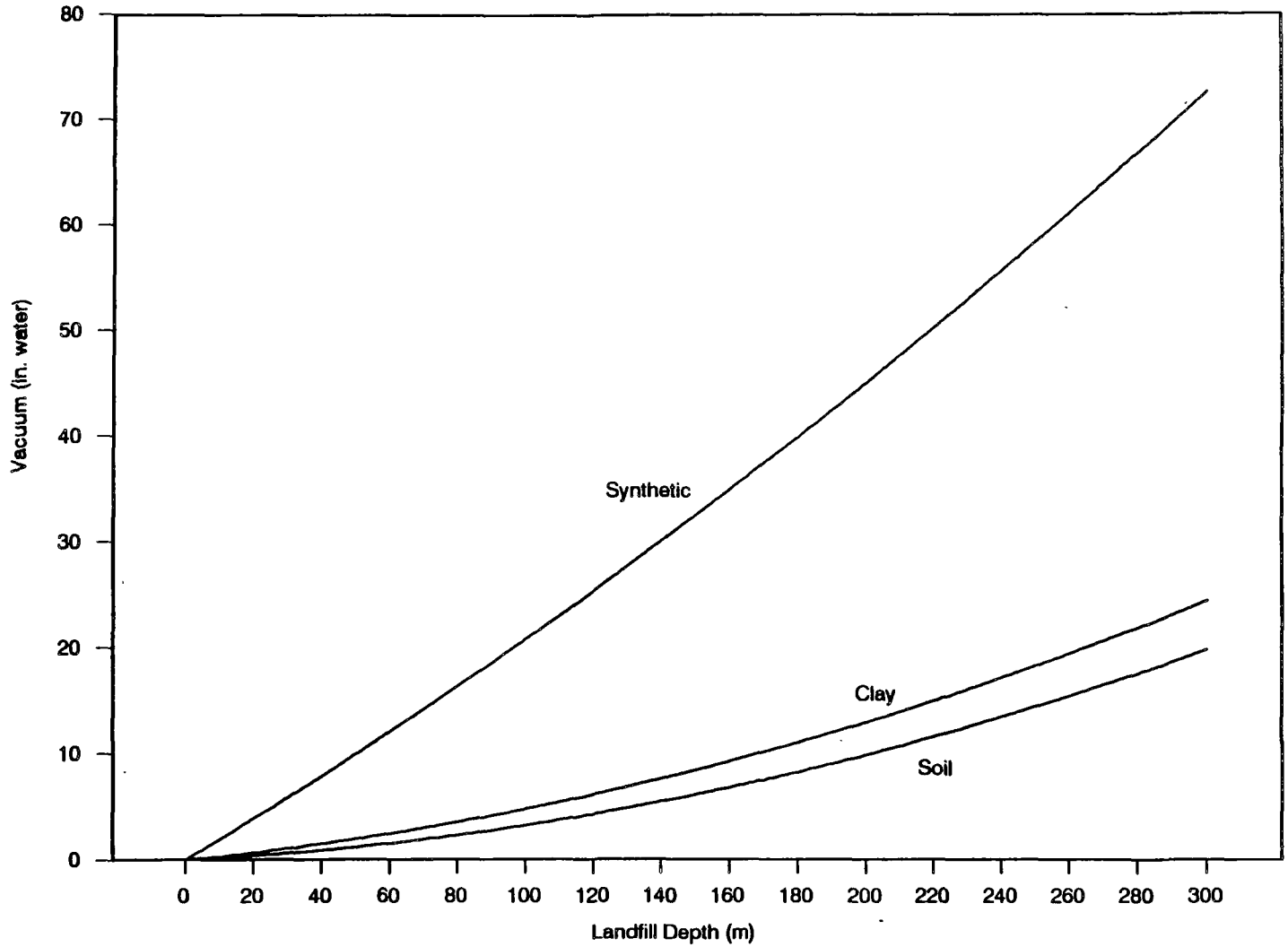

Figure 9-6. Maximum Blower Vacuum as a Function of Landfill Depth for Three Cover Types

230 m^3/Mg. The theoretical basis for Figure 9-6 is further described in Appendix G.

In cases where field measurement is not performed, the proposed well vacuum should be compared to the predicted maximum from Figure 9-6. If the proposed vacuum is less than or equal to that indicated in Figure 9-6, then the proposed value can be used to determine the radius of influence from Figure 9-7. If the proposed well vacuum is greater than the maximum indicated in Figure 9-6, then the value obtained from Figure 9-6 should be used to determine the radius of influence from Figure 9-7. Consistent with the theoretical correlation presented for maximum well vacuum, the correlation presented in Figure 9-7 for radius of influence is based on the Scholl-Canyon model with a k of .02 $years^{-1}$ and L_0 of 230 m^3/Mg. The theoretical basis and calculations are detailed in Appendix G.

Once the radius of influence is determined, the proposed well placement can be evaluated. Identical to the criteria outlined above when using a field measured radius of influence, the wells are to be sited along the perimeter of the landfill no more than the radius of influence from the landfill perimeter and two times the radius of influence apart. Once the perimeter wells are sited, then wells are to be sited throughout the interior of the landfill, at a distance of no more than two times the radius of influence. The only difference in this technique and the one described above is that a single radius of influence is used in siting both perimeter and interior wells.

9.2.1.2 Well Construction. The landfill gas extraction well is to be constructed of polyvinyl chloride (PVC), high density polyethylene (HDPE) pipe, fiberglass, stainless steel, or other suitable nonporous material, at least 3 inches in diameter. The well should extend from the landfill surface to at least 75 percent of the landfill depth. It is recommended that the bottom two thirds of the pipe be perforated with 1/2 inch diameter holes spaced at 90 degrees every 6 inches. Slotted pipe having equivalent perforations is also suitable. The pipe should be placed in the center of a 2 ft diameter bore and backfilled with gravel to a level 1 ft above the perforated section. A 4 ft layer of backfill material should be placed on top of the gravel followed by at least 3 ft of bentonite. The remainder of

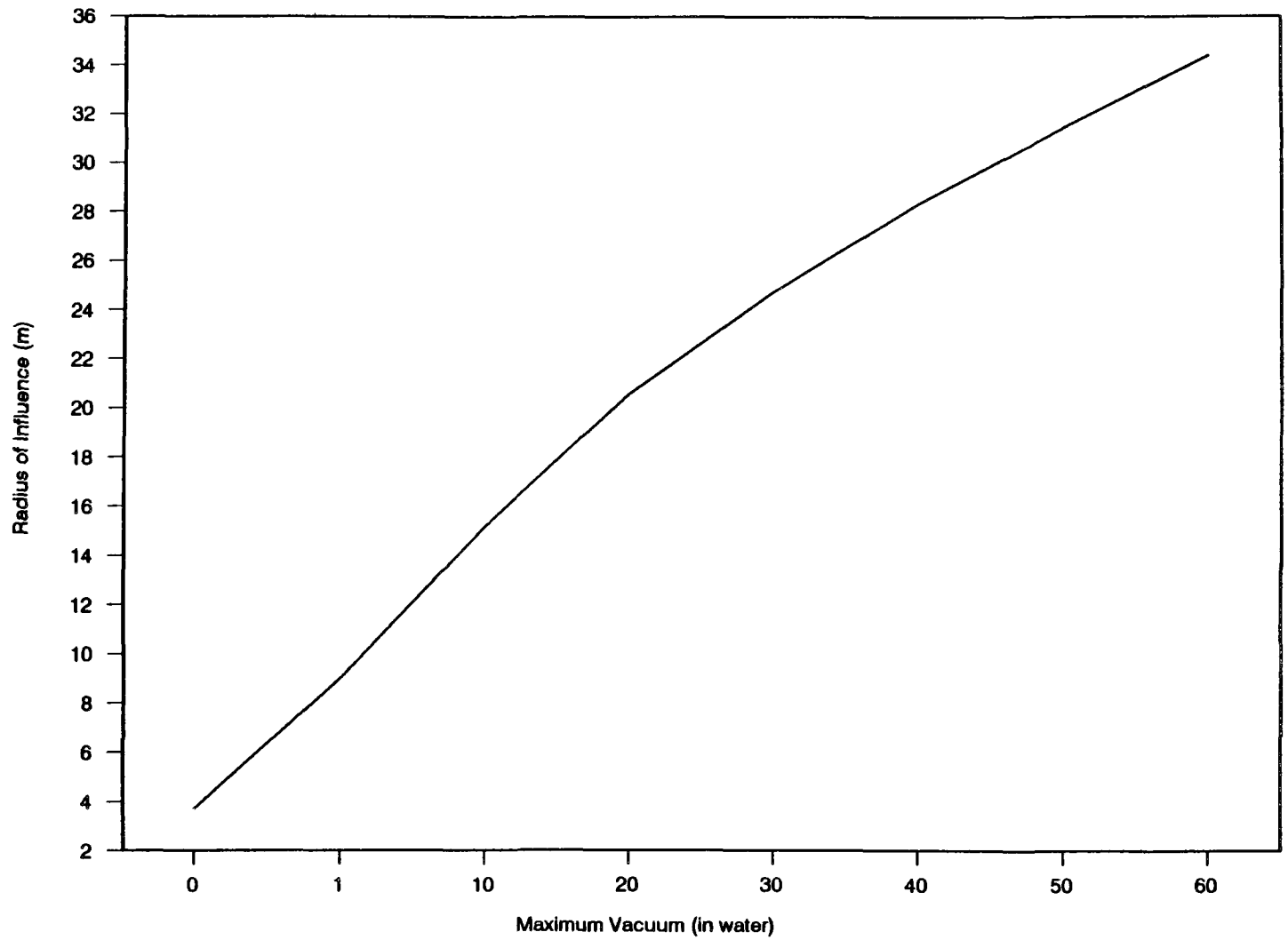

Figure 9-7. Estimated Radius of Influence as a Function of blower Vacuum

the bore can be backfilled with cover material or a material of equal or lower permeability.

9.2.1.3 Monitoring and Adjustment Design Provisions. To facilitate periodic well monitoring and adjustment, the well head should be equipped with a valve, flanges, gaskets, connectors and access couplings. The well assembly should also include at least one sample port that can be used to monitor pressure or collect gas samples periodically. The extraction well assembly and well head assembly are illustrated in Figure 9-8.

The well head may be connected to the collection header pipes below or above the landfill surface. The advantage of installing header pipes above ground is the ease of maintenance and operation. The disadvantage is the higher probability of damaging header pipes with landfill operating equipment and the possibility of blockage in the pipeline due to the condensate freezing in areas with severe winters.

9.2.1.4 Gas Mover Sizing. The gas mover (fan, blower or compressor) system should be designed to handle the peak landfill gas flowrate over the life of the gas moving equipment. This attribute can be evaluated by first projecting the peak landfill gas flowrate and comparing this flow to the proposed equipment specifications. The peak gas flow rate can be projected using the following expression:

$$\text{Peak Flow } [m^3/yr] = 2L_0 \, R \, (1 - e^{-kt})$$

where,

L_0 = refuse methane generation potential, m^3/Mg refuse

R = average annual acceptance rate, Mg/yr

k = methane generation rate constant, 1/yr

t = age of the landfill plus the gas mover equipment life or active life of the landfill, which ever is less, in years

A value of 230 m^3/Mg is recommended for L_0. If Method 2E has been performed, the value of k determined from the test should be used; if not, a value of .02 $years^{-1}$ is recommended.

9.2.2 Design Guidelines for Active Horizontal Collection Systems

Four design features of the proposed horizontal collection system should be evaluated by the State reviewer to ensure that an effective

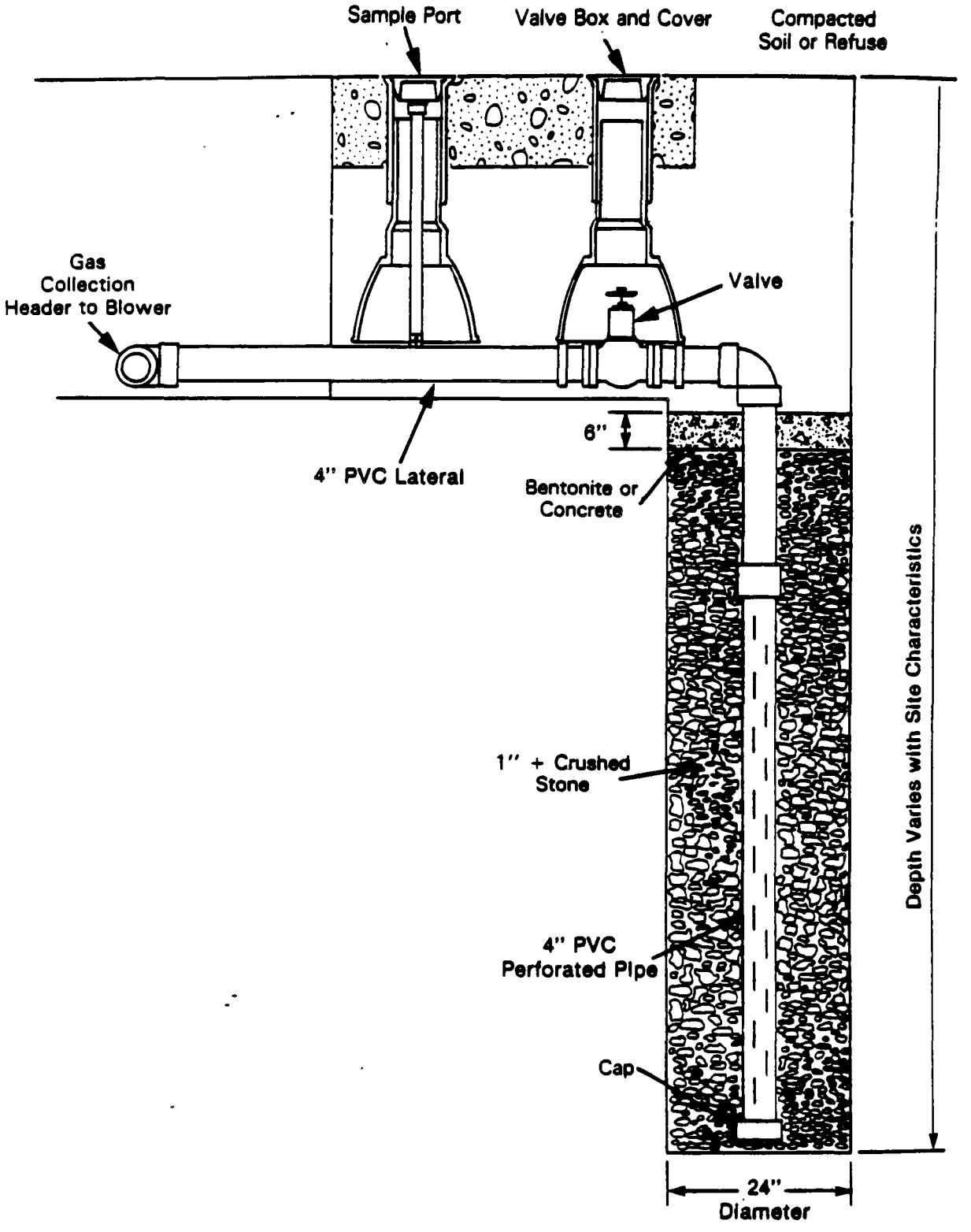

Figure 9-8. Gas extraction well and well head assembly.

collection system is installed. These are the proposed well spacing, the proposed trench construction, provisions for trench monitoring and adjustment, and capacity of the gas mover system. Each of these design features are addressed below.

9.2.2.1 Horizontal Trench Spacing. The preferred method for determining effective trench spacing at a specific landfill is the use of field measurement data. Although EPA Method 2E is based on a vertical well test, results of this method can be used to determine radius of influence in the horizontal direction. If such a determination has been made using EPA Method 2E, the determined radius of influence is to be used in setting the horizontal spacing. The trenches should be spaced at a distance of no more than two times the measured radius of influence (measured radius of influence for internal vertical wells) apart. The vertical spacing of trenches, however should be closer. Since compaction of the refuse causes refuse permeability to be lower in the vertical direction, influence of the trench is less in the vertical direction than in the horizontal direction. A vertical spacing of one forth the horizontal spacing is recommended to account for lower permeability in the vertical direction.

In situations where the landfill owner chooses not to perform EPA Method 2E, the well spacing is to be determined based on the same theoretical concepts presented in Section 9.2.1.1 for vertical well spacing. Using the proposed trench vacuum, the theoretical radius of influence in the horizontal direction can be obtained from Figure 9-7. This radius of influence is to be used identically to the interior radius of influence determined discussed above. The trenches are to be spaced no more than two times the theoretical radius of influence apart horizontally, and vertically no more than one-half the theoretical radius of influence.

9.2.2.2 Trench Construction. The horizontal trenches may be constructed of PVC, HDPE, corrugated steel, or other suitable nonporous material. In order to minimize the collapse of the trenches due to the refuse accumulation and/or landfill operation equipment, some employ alternating pipe connections which typically consist of pipes with adjacent diameters (e.g., 8" and 10", 10" and 12", etc.) loosely fitted together. Loose fitting pipes of different diameters allow landfill gas to freely flow

through yet also handles the stress due to the refuse weight and/or equipment better than straight pipe connections. Some landfill owners prefer using corrugated steel pipes since the heat of the landfill tends to reduce the stress strength of PVC or HDPE pipes. Typical construction of the horizontal trench collection system is illustrated in Figure 9-9.

9.2.2.3 Monitoring and Adjustment Design Provisions. To facilitate periodic trench monitoring and adjustment, each layer of trenches should be connected to a common header leg that extends to the surface and is equipped with a valve, flanges, gaskets, connectors and access couplings. The header leg assembly should also include at least one sample port that can be used to monitor pressure or collect gas samples periodically. The trench header assembly should allow for controlling individual layers of trenches.

9.2.1.4 Gas Mover Sizing. The gas mover (fan, blower or compressor) system should be designed to handle the peak landfill gas flowrate over the life of the gas moving equipment. Identical to vertical well collection systems, this attribute can be evaluated by first projecting the peak landfill gas flowrate and comparing this flow to the proposed equipment specifications. The peak gas flow rate can be projected using the following expression:

$$\text{Peak Flow } [m^3/yr] = 2L_0 \, R \, (1 - e^{-kt})$$

where,

L_0 = refuse methane generation potential, m^3/Mg refuse

R = average annual acceptance rate, Mg/yr

k = methane generation rate constant, 1/yr

t = age of the landfill plus the gas mover equip. life or active life of the landfill, which ever is less, in years

A value of 230 m^3 is recommended for L_0. If Method 2E has been performed, the value of k determined from the test should be used; if not, a value of .02 $years^{-1}$ is recommended.

9.2.3 Design of Passive Collection Systems

As indicated above, passive systems are accepted as BDT only when combined with a synthetic liner on the top, bottom, and sides of the

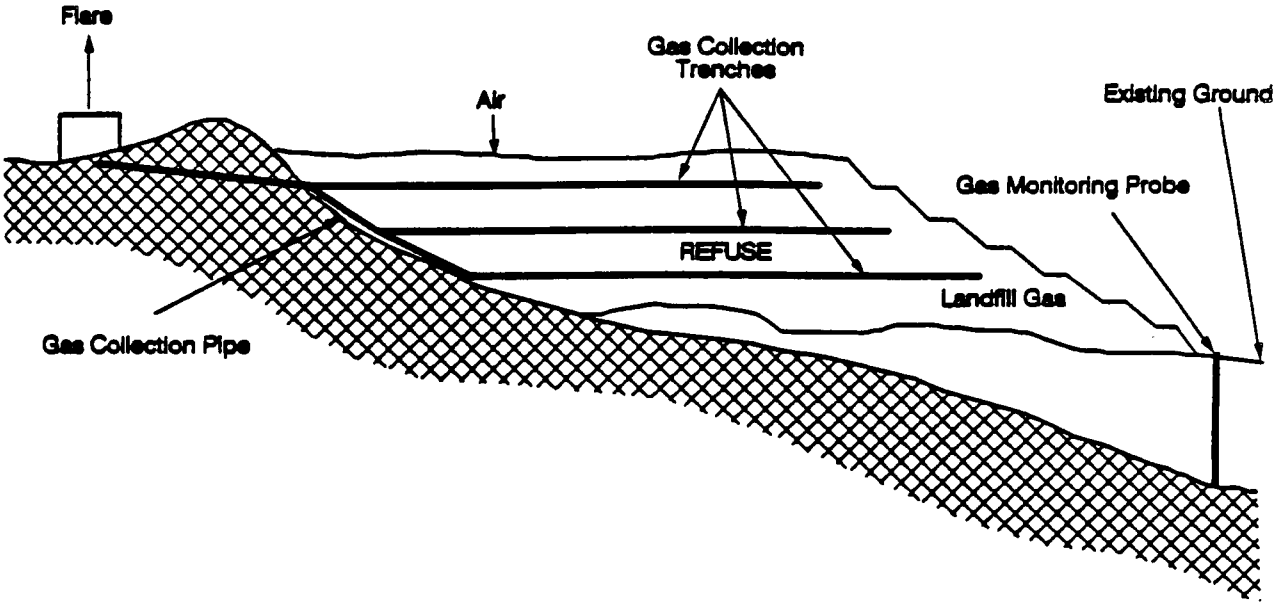

Figure 9-9. Horizontal trench collection system.

landfill. If such a collection system is proposed, two design features will need to be evaluated, the proposed well spacing and the proposed well construction. Each of these design features are addressed below.

9.2.3.1 Passive Well Spacing. The preferred methodology for determining the well spacing for passive collection systems is to use the average static landfill pressure determined from field testing. If EPA Method 2E has been performed, first determine the average static landfill pressure using all of the deep probe static pressure measurements. Second, the pressure drop across the control system should be established, based on control equipment specifications. The pressure drop across the flare (or other control device), flame arrester, and collection header piping should be considered. The expected pressure drop across the control system (usually provided in vendor specifications) should be subtracted from the landfill pressure to determine the differential pressure driving force. Using this differential pressure (between the landfill gauge pressure and the control system pressure drop), the theoretical radius of influence can be determined using Figure 9-10. Based on this theoretical radius of influence, wells should be placed throughout the landfill such that all areas of the landfill are covered and the distance between wells is no more than two times the radius of influence.

If EPA Method 2E has not been performed at the landfill, then the static landfill pressure should be determined by field measurement. The landfill should be divided into 5 equal volumes of refuse and a pressure probe should be installed near the center of each equal volume, following the probe installation procedures outlined in Section 3.3.1 of EPA Method 2E. A differential pressure gauge should be used to measure the gauge pressure at each pressure probe every 8 hours for 3 days. All 120 of these pressure measurements should be averaged to determine the static landfill pressure. This static landfill pressure should be used the same as Method 2E results (discussed above). The expected control system pressure drop (including the flare tip, flame arrester, collection header) is to be subtracted from the static landfill pressure to determine the differential pressure driving force. This differential pressure can then be used in conjunction with Figure 9-10 to determine the theoretical radius of

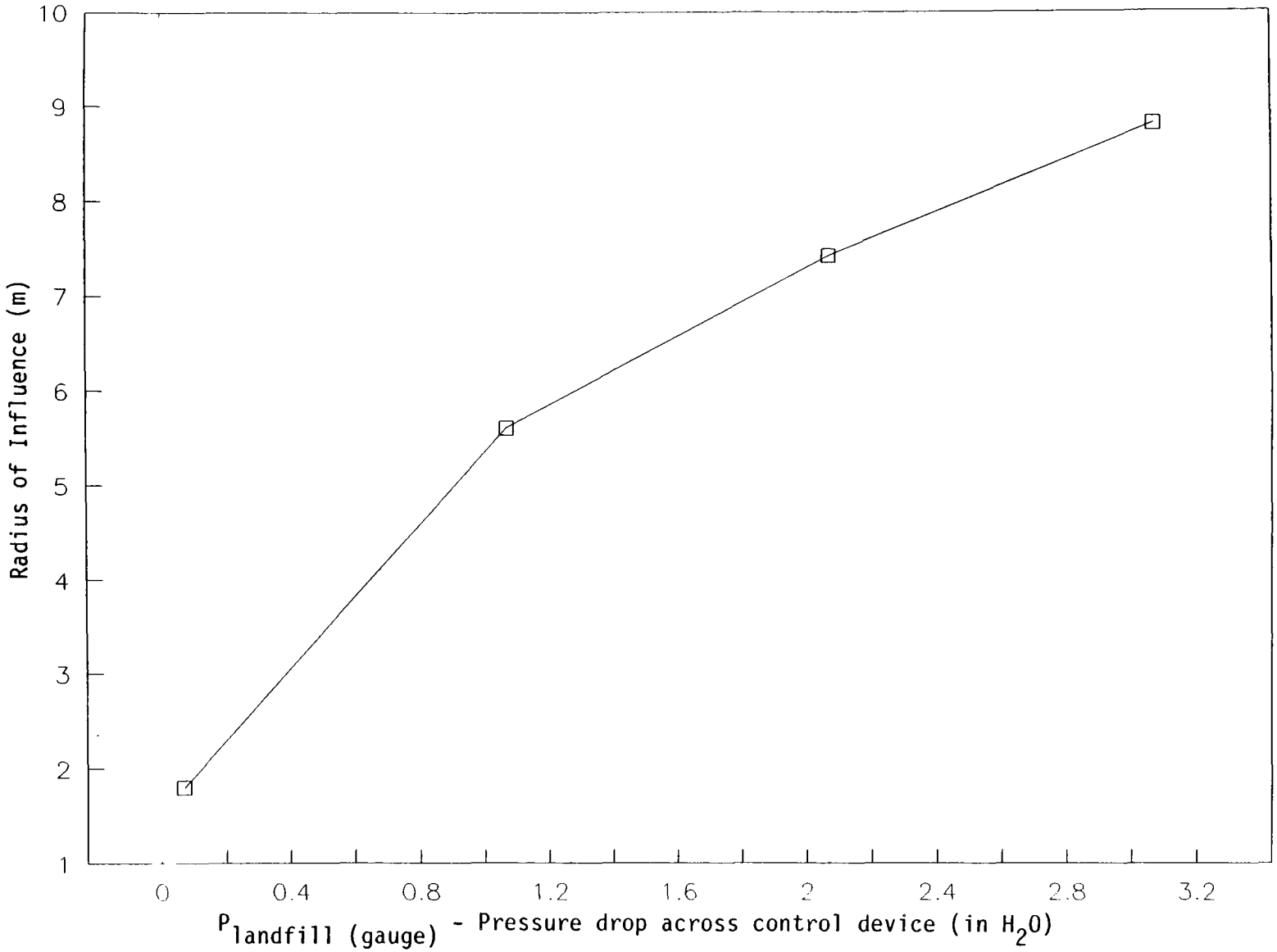

Figure 9-10. Estimated radius of influence for a passive system well as a function of the collection/control system pressure drop.

influence. Wells should be placed throughout the landfill such that all areas of the landfill are covered and the distance between wells is no more than two times the radius of influence.

9.2.3.2 Passive Well Construction. The passive extraction well is to be constructed of polyvinyl chloride (PVC) or high density polyethylene (HDPE) pipe, at least 4 inches in diameter. The well should extend from the landfill surface to at least 75 percent of the landfill depth. It is recommended that the bottom two thirds of the pipe be perforated with 1/2 inch diameter holes spaced at 90 degrees every 6 inches. The pipe should be placed in the center of a 2 ft diameter bore and backfilled with gravel to a level 1 ft above the perforated section. The remainder of the hole should be backfilled with a cover or backfilling material.

The well construction for passive systems is much less critical than active systems. This is primarily because the collection well is under positive pressure and air infiltration is not a concern. Additionally, elaborate well head assemblies are not required since monitoring and adjustment is not necessary. However, it is important that a good seal be provided around the passive well in order to maintain the integrity of the synthetic liner and maximize containment. Therefore, it is recommended that a boot type seal, flange type seal, concrete mooring or other sealing technique be used at each well location to maintain integrity of the landfill cap.

9.3 COLLECTION SYSTEM OPERATING GUIDELINES

Active landfill gas collection systems should be periodically monitored and adjusted to: (1) maximize landfill gas collection, and (2) ensure that air infiltration into the system does not exceed safe levels. Additionally, due to the inconsistency typically found within landfills, it may be necessary to install additional wells in certain areas of high gas generation.

To insure effective collection of landfill gas, the pressure and air content should be measured at each well head (vertical collection systems) or common header leg (horizontal collection systems) at least once every month. If the measured pressure at the well head is positive, then the flow from that well or set of trenches should be increased by opening the valve.

Infiltration of too much air into a landfill may cause a fire or explosion hazard. Therefore, EPA has determined that the N_2 concentration (as a surrogate for air concentration) in the collected gas should be maintained under 1 percent by volume. If the N_2 concentration exceeds 1 percent, the valve at the well head assembly should be adjusted to decrease the flow from that well, thus decreasing the level of air infiltration. In cases where the well or leg pressure is positive and the flow cannot be increased due to the exceedance of the N_2 concentration limit, additional extraction wells should be installed and added to the collection system.

In all types of collection systems with header piping, condensation of water and organics is expected to occur as a result of cooler temperatures above the surface of the landfill. This condensate is generally collected, treated for pH, and routed to a water treatment facility or discharged under NPDES permit or otherwise handled according to RCRA Subtitle D and/or Subtitle C requirements.

9.4 DESIGN AND OPERATING GUIDELINES FOR CONTROL SYSTEMS

All collected landfill gas must be routed to a control device capable of achieving 98 percent reduction of the NMOC emissions by weight. The Agency has identified a number of control devices that can achieve the specified reduction. These include: open flares, enclosed ground flares, gas turbines, internal combustion (IC) engines, boilers, incinerators, and purification systems. Open flares that are in conformance with the design and operating requirements of 40 CFR 60.18 are assumed to yield 98 percent destruction of NMOC emissions. Enclosed combustors, however, such as enclosed ground flares, turbines, IC engines, boilers, and incinerators, require a performance test to demonstrate 98 percent destruction efficiency or an outlet NMOC concentration of 20 ppmvd at 3 percent oxygen using EPA Method 25. Purification systems, such as adsorption and absorption, do not require performance testing if all vent streams from the system are routed to an open flare or enclosed combustor that meet the specifications listed above. Control of only some portion of the vent streams would be allowed if overall 98 percent destruction in NMOC emissions is achieved.

Alternatively, the landfill owner may select any NMOC destruction device, or design and operate one of the listed devices outside the range of the parameters specified if the device can be demonstrated to achieve 98 percent destruction of NMOC emissions. EPA Method 25 should be used to determine the performance of alternative control devices.

9.5 COMPLIANCE SCHEDULE

Landfill owners/operators of all designated existing MSW landfills are required to submit a design capacity report and an initial NMOC mass emission rate estimate (Tier 1) within 90 days of the effective data of their respective approved State plan for implementing the emission and compliance guidelines. Owners and/or operators of new landfills must submit a design capacity report and an initial NMOC mass emission rate estimate (Tier 1) within 90 days of start-up (i.e., refuse acceptance). Suggested contents of the report are discussed in Section 9.1.

Landfills with design capacities less than 100,000 Mg are not required to perform further testing or reporting, unless the design capacity is changed due to the addition of new areas, increase in depth, etc. If such a change occurs, the landfill owner/operator is required to submit an amended design capacity report within 90 days of the change.

Landfills with design capacities greater than 100,000 Mg, must file an annual or periodic report of the NMOC mass emission rate (Tier 1) until the landfill closes or the rate exceeds the regulatory cutoff.

When the NMOC emission rate, calculated in Tier 1, reaches 150 Mg/yr, the owner/operator must submit either a notification of intent to install a collection system based on the specifications in Section 60.758 or a collection system design plan for review within 1 year. If the landfill owner/operator elects to perform the Tier 2 sampling in order to generate a site-specific NMOC concentration or gas generation rate to use for the calculation of the more precise NMOC emission rate, he/she must report these calculations within one year of the initial Tier 1 calculation as well.

If the NMOC emission rate calculated in Tier 2 equals or exceeds 150 Mg/yr, then either controls must be installed or the owner/operator can choose to perform Tier 3 testing; either must be done within 1 year after

agency approval of a design which has been submitted for review, which takes approximately 6 months, or within 18 months after the submittal of a notification of intent. Should the NMOC emission rate calculated in Tier 2 be below 150 Mg/yr, then the Tier 2 calculation must be repeated annually, while updating the NMOC concentration data at the specified intervals, as described in Section 9.1. If the value for the NMOC emission rate from the Tier 3 testing still equals or exceeds 150 Mg/yr then controls must be installed within one year of the Tier 3 results. If the Tier 3 emission rate calculation is below 150 Mg/yr then the Tier 3 calculation must be repeated annually, while updating the NMOC concentration data at the specified intervals, as described in Section 9.1.

The Tier 3 test will be valuable for those landfills that need to install collection systems, because, as discussed in Section 9.2, flow rates obtained may be used in designing the collection system. Additionally, the test wells can serve as collection wells, if they meet the operating criteria.

After the collection and control systems have been installed, the owner/operator has 90 days to complete and submit the initial performance test results. Also, semiannual compliance reports must be submitted in which the following would be included: (1) any period in which the value of any of the monitored operating parameters falls outside the ranges identified in the initial performance test; (2) results of all annual performance tests; (3) identification of any periods for which data were excluded from these calculations; (4) any period when air pollution control equipment malfunction occurred.

Upon closure of the landfill, a closure report must be filed. If, after closure, the landfill meets the criteria outlined in Section 9.1 for discontinuing control, the landfill owner/operator must submit a report. The report should include documentation verifying that the collection and control system has been operating according to the specifications for a minimum of 15 years and that the NMOC mass emission rate has been below 150 Mg/yr for three consecutive 90 day-periods.

The landfill owner/operator may discontinue control upon the State's verification that the above requirements have been met.

The proposed regulation would also require that certain types of records be maintained. Records of the accumulated refuse in place, collection system design (including proposed and subsequent well or trench spacing), control device vendor specifications, the initial performance test results, and monitoring parameter established during the initial performance test, must be maintained on site as long as the collection system and control devices are required to be operated.

9.6 REFERENCES

1. Y.C. McGuinn, Radian Corporation, to S.A. Thorneloe, EPA:CPB, February 22, 1989, Design of municipal solid waste landfill gas collection systems and their relative installation costs.

APPENDIX A

EVOLUTION OF THE BACKGROUND INFORMATION DOCUMENT

APPENDIX A

EVOLUTION OF THE BACKGROUND INFORMATION DOCUMENT

A.1 INTRODUCTION

The purpose of this study was to develop background information to support New Source Performance Standards (NSPS) for Municipal Solid Waste Landfills (MSW landfills). Work on this study was performed by the Radian Corporation from August 1987 to _____ 1990 under contract with the U. S. Environmental Protection Agency, Office of Air Quality Planning and Standards.

The following chronology lists the major events which have occurred during the development of background information for the MSW landfills NSPS Major events are divided into three categories: (1) site visits, (2) meetings and briefings, (3) reports and mailings.

G.2 SITE VISITS

November 16, 1987	Site visit to Puente Hills Landfill, Whittier, CA
November 17, 1987	Site visit to Toyon Canyon Landfill Power Station, Los Angeles, CA
November 18, 1987	Site visit to Palos Verdes Landfill, Whittier, CA
November 18, 1987	Site visit to Rossman Landfill, Oregon City, OR
December 15, 1987	Site visit to Rumpke Landfill, Greensboro, NC
September 13, 1989	Site visit to Wilder's Grove Landfill, Raleigh, NC

G.2 MEETINGS AND BRIEFINGS

November 16, 1987	Meeting with representatives of the Los Angeles County Sanitation District
November 17, 1987	Meeting with representatives of the South Coast Air Quality Management District

March 21-24, 1988	Presentation at Governmental Refuse Collection and Disposal Association symposium, Houston, TX
May 17, 1988	Meeting with representatives of the Governmental Refuse Collection and Disposal Association to discuss comments on draft background information document
May 18-19, 1988	Presentation at the National Air Pollution Control Techniques Advisory Committee (NAPCTAC)
June 8, 1988	Meeting with representatives of Waste Management, Inc., to discuss comments on draft background information document
August 24, 1988	Meeting with Waste Management of North America, Inc. and the landfill Gas Committee of the Governmental Refuse Collection and Disposal Association, to discuss comments on draft background information document
October 5, 1988	Meeting with representatives of Browning-Ferris Industries to discuss status of project
January 19, 1989	Meeting with Browning-Ferris Industries to discuss responses to Section 114 letters
March 16, 1989	Meeting with Waste Management, Inc. to discuss status of project and Section 114 responses
March 20-24, 1989	Presentation of status of project at Governmental Refuse Collection and Disposal Association symposium, Monterey, CA
May 4, 1989	Presentation of status of project at National Solid Waste Management Association (NSWMA) in Chicago, IL
June 7, 1989	Presentation at the National Air Pollution Control Techniques Advisory Committee (NAPCTAC)
September 6, 1989	Meeting with representatives of Combustion Engineering to discuss comments on field test procedures

G.3 REPORTS AND MAILINGS

April 5, 1988	Mailing for NAPCTAC meeting on May 18, 1988

March 15, 1989	Mailing for public comment on the preliminary analysis of the design and costing for collection systems
July 14, 1989	Mailing for public comment on draft field test procedures and test methods

APPENDIX B

INDEX TO ENVIRONMENTAL CONSIDERATIONS

APPENDIX B

INDEX TO ENVIRONMENTAL CONSIDERATIONS

A.1 INTRODUCTION

This appendix consists of a reference system which is cross indexed with the October 21, 1974, Federal Register (30 FR 37419) containing EPA guidelines for the preparation of Environmental Impact Statements. This index can be used to identify sections of the document which contain data and information germane to any portion of the Federal Register guidelines.

The are, however, other documents and docket entries which also contain data and information, of both a policy and a technical nature, used in developing the proposed standards. This appendix specifies only the portions of this document that are relevant to the indexed items.

TABLE B-1. INDEX TO ENVIRONMENTAL CONSIDERATIONS

Agency guideline for preparing regulatory action environmental impact statements (39 FR 37419)	Location within the background information document
• Background and Summary of <u>Regulatory Alternatives</u>	
- Regulatory alternatives	The regulatory alternatives are summarized in Chapter 5.
- Statutory basis for proposing standards	The statutory basis for the proposed standards is summarized in Chapter 1.
- Source category and affected industries	A discussion of the source category is in Chapter 3; details of the "business/economic" nature of the industries affected are presented in Chapter 8. Affected are presented in Chapter 8.
- Emission control technologies	A discussion of emission control technologies is presented in Chapter 4.
• Environmental, Energy, and Economic Impacts of <u>Regulatory Alternatives</u>	
- Regulatory alternatives	Various regulatory alternatives are discussed in Chapter 5.
- Environmental impacts	The environmental impacts of various regulatory alternatives are presented in Chapter 6, Section 6.1, and 6.2.
- Energy impacts	The energy impacts of various regulatory alternatives are presented in Chapter 6, Section 6.3.
- Cost impacts	Cost impacts of various regulatory alternatives are presented in Chapter 7.
- Economic impacts	The economic impacts of various regulatory alternatives are presented in Chapter 8.

APPENDIX C
LANDFILL GAS COMPOSITION DATA

APPENDIX C

LANDFILL GAS COMPOSITION DATA

The speciated landfill gas composition data for 46 municipal solid waste landfills are presented in Table C-1. This data was obtained from Section 114 responses and South Coast Air Quality Management District Test Reports. The identity of the landfills evaluated have been withheld due to the presence of confidential business information. All of the data is reported in ppmv unless otherwise noted.

TABLE C-1. SPECIATED NMOC COMPOSITION

LANDFILL ID	A	B	C	D	E	F	G	H	I	J	K	L	M	N
CHEMICAL NAME														
ETHANE	929.5				1780		0	0		0	0	0	0	0
TOLUENE	35.53	758	49.3	244.5	60.91	0.251	19.38	77.17		3	17.2	1.45	12.7	28.22
METHYLENE CHLORIDE	0.18		174		38.82		44	14	1.8	8	0.7	0.2	35	3.25
HYDROGEN SULFIDE														
ETHYLBENZENE	36.95	428			4.2		0.25	7		0.15	1.3	0.23	0.65	4.06
XYLENE		664												
1,2 - DIMETHYL BENZENE		588												
LIMONENE		470												
TOTAL XYLENE ISOMERS	64.98				2.47		0.5	15.28		0.45	2.9	9.78	1.55	8.55
α-PINENE		446												
DICHLORODIFLUOROMETHANE	0	19			43.99		32.95	11.92		23.3	0	11.85	34.5	1.3
ETHYLESTER BUTANOIC ACID		398												
PROPANE	9.76				48.8		0	4.67		7	5.2	0	6.5	0
TETRACHLOROETHENE	0.74	77		28.5	14.93	0.177	7.1	5.63	0.1	0.9	0.3	0.23	1	2.4
VINYL CHLORIDE	9.98		0.05	48.1	6.11	1	15	10.92		11.35	1	2.7	7.7	8.43
METHYLESTER BUTANOIC ACID		305												
ETHYLESTER ACETIC ACID		282												
PROPYLESTER BUTANOIC ACID		253												
1,2 - DICHLOROETHENE	0.13	34		84.7	8.58		2.78	7.82		1.65	0	0.83	1.2	5.27
METHYL ETHYL KETONE	0				1.48		0	7.67		0	0	3.75	3.65	12
THIOBISMETHANE		210												
METHLYCYCLOHEXANE		197												
TRICHLOROETHENE	0.22	34		20.4	6.96	0.069	1.38	5.23	0.8	0.4	0.3	0.18	1.55	1.54
NONANE		167												
BENZENE	1.53	23	0.95	52.2	2.76	0.299	1.05	1.53	0.4	0.1	0.3	0.35	0.7	2.6
ETHANOL		157												

TABLE C-1. SPECIATED NMOC COMPOSITION

LANDFILL ID	O	P	Q	R	S	T	U	V	W	X	Y	Z	AA	BB
CHEMICAL NAME														
ETHANE	0		268.75	0		1420	0	0			0	0	0	
TOLUENE	40		125.28	37	13	221	13.9	5.85	0.197	34.2	68.5	30	2.5	
METHYLENE CHLORIDE	127.5	0.00536	29.91	14	0.5	24.5	24.67	2	0.146666		3.45	50	2	18.39
HYDROGEN SULFIDE					700									
ETHYLBENZENE	5		35.35	4	3.4	48.1	3.73	0.7			22	3.8	0.55	
XYLENE														
1,2 - DIMETHYL BENZENE														
LIMONENE														
TOTAL XYLENE ISOMERS	12.5		70.75	12		0	4.63	1.5			67.5	12	1.3	
α -PINENE														
DICHLORODIFLUOROMETHANE	7.45		16	0	0	0	24.47	11.45			16.5	30	0.5	
ETHYLESTER BUTANOIC ACID														
PROPANE	86.5		4.26	0		18.2	1.4	11			0	68	0	
TETRACHLOROETHENE	11.95		12.63	11		8.2	2.63	0.4	0.0035	5.4	7.75	9.3	0.4	12.13
VINYL CHLORIDE	19	2	16.92	13	0.84	15.2	12.43	5.2	0.7	3.42	3	5.3	0.4	2.65
METHYLESTER BUTANOIC ACID														
ETHYLESTER ACETIC ACID														
PROPYLESTER BUTANOIC ACID														
1,2 - DICHLOROETHENE	18.5		4.55	13	6.5	0	3.93	0.5		0.016	1.35	0.9	0.25	
METHYL ETHYL KETONE	4.95		18.75	5.5		NM	5	6			57.5	15	NM	
THIOBISMETHANE														
METHLYCYCLOHEXANE														
TRICHLOROETHENE	21.5	0.00615	12.98	3.1	0.2	7.85	1.67	0.2	0.0158	4.86	4.7	3.4	0.2	1.14
NONANE														
BENZENE	1.95	0.00436	5.53	1.2	0.57	2.42	0.77	0.15	0.186666	1.48	1.5	1	0	1.04
ETHANOL														

TABLE C-1. SPECIATED NMOC COMPOSITION

LANDFILL ID	CC	DD	EE	FF	GG	HH	II	JJ	KK	LL	MM	NN	OO	PP
CHEMICAL NAME														
ETHANE		0	0	0		0			0					0
TOLUENE		47.5	2.1	27.2	31.5	23.33	8.63		53	64		4.73	15	10.05
METHYLENE CHLORIDE	82	9.25	3	0	20	0.33			0	54.9	18.4		32	17
HYDROGEN SULFIDE										11	47.9			
ETHYLBENZENE		10.9	0.2	2.73	5.7	5.27			4.6	1.7			2.2	0.3
XYLENE													3.7	
1,2 - DIMETHYL BENZENE														
LIMONENE														
TOTAL XYLENE ISOMERS		37.5	0.45	5.57	10	13.33			12					0.75
α -PINENE														
DICHLORODIFLUOROMETHANE		8.85	14.25	8.9	11.75	13.27			19	0	7.3			37.5
ETHYLESTER BUTANOIC ACID														
PROPANE		0	6.5	0.63		0			0					36.5
TETRACHLOROETHENE		12.25	0.25	1.53	4.6	3.7			3.8		7.5	0.012	1	0.95
VINYL CHLORIDE	6.7	7.6	1.95	14.4	2.05	4.93	18.73		0	4.5	7.7	3.43		3.25
METHYLESTER BUTANOIC ACID														
ETHYLESTER ACETIC ACID														
PROPYLESTER BUTANOIC ACID														
1,2 - DICHLOROETHENE		5.45	0.3	2.87	6.2	6.23			8.8	3.8	0	0.097	1.2	0.9
METHYL ETHYL KETONE		11	NM	6.33	5	31.33			21					4.7
THIOBISMETHANE														
METHLYCYCLOHEXANE													2.4	
TRICHLOROETHENE		3.75	0.15	0.5	3.25	1.63	0.76	9.47	1.8	1.2	3.9	0.025	2.4	0.45
NONANE														
BENZENE	4	0.65	0	0.83	1	0.57	0.916	32.3	0.6	0.77		2.84	1.2	0.2
ETHANOL														

TABLE C-1. SPECIATED NMOC COMPOSITION

LANDFILL ID / CHEMICAL NAME	QQ	RR	SS	TT
ETHANE			930	1240
TOLUENE	8.65	4.91	123	51
METHYLENE CHLORIDE			1.48	50.95
HYDROGEN SULFIDE				
ETHYLBENZENE			23.4	7.22
XYLENE				
1,2 - DIMETHYL BENZENE				
LIMONENE				
TOTAL XYLENE ISOMERS			70.9	22.8
α-PINENE				
DICHLORODIFLUOROMETHANE			0	0.19
ETHYLESTER BUTANOIC ACID				
PROPANE			13.1	25.3
TETRACHLOROETHENE	0.3017	0.441	6.82	64.95
VINYL CHLORIDE	14.28	2.57	5.61	3.83
METHYLESTER BUTANOIC ACID				
ETHYLESTER ACETIC ACID				
PROPYLESTER BUTANOIC ACID				
1,2 - DICHLOROETHENE	0.1638	0.28	0.11	1.3
METHYL ETHYL KETONE				
THIOBISMETHANE				
METHLYCYCLOHEXANE				
TRICHLOROETHENE	0.309	0.748	2.02	7.8
NONANE				
BENZENE	0.595	2.57	2.65	4.55
ETHANOL				

TABLE C-1. SPECIATED NMOC COMPOSITION

LANDFILL ID	A	B	C	D	E	F	G	H	I	J	K	L	M	N
CHEMICAL NAME														
ACETONE	0				1.84		2.25	4.5		0	0	0	2.5	2.25
2 - BUTANOL		152												
OCTANE		152												
PENTANE	0.58				11.1		0	3.83		0.5	1.2	0	9	0
HEXANE	2.49				20.82		0	4.17		3	2.4	0	10	0
METHYLESTER ACETIC ACID		136												
1 - METHOXY - 2 - METHYL PROPANE		136												
2 - BUTANONE		129												
1,1 - DICHLOROETHANE	0.3				11.85		11.18	5.63		1.75	0.6	0.05	0	0.85
1 - BUTANOL		100												
BUTANE	0				18.76		0	0.83		1	1	0	5	0
4 - METHYL - 2 - PENTANONE		89												
2 - METHYL PROPANE		84												
1 - METHYLETHYLESTER BUTANOIC ACID		69												
2 - METHYL, METHYLESTER PROPANOIC ACID		69												
CARBON TETRACHLORIDE	0			0.065	0	0.0026	0	0		0	0	0.05	0	0
CHLOROETHANE	0.43				3.25		9.2	2.33		1.6	0	0.5	8.25	0.2
1,1,3 TRIMETHYL CYCLOHEXANE		57												
2 - METHYL - 1 - PROPANOL		51												
1,2 - DICHLOROETHANE	0.02			30.1	0.02	0.447	0.78	0	0.05	0	0	0	0	0.55
TRICHLOROFLUOROMETHANE	0.66	0	0	0	1.35	0	1.08	1.3	0	2.35	0.7	0.73	7.9	0.48
CHLOROMETHANE	1.12				0.9		0.28	0.18		1.25	0	0	6.1	0.1
2,5 DIMETHYL FURAN		41												
2 - METHYL FURAN		40												
CHLORODIFLUOROMETHANE	0.97				12.58		0	0.77		3.85	0	0	3	0
PROPENE		36												

TABLE C-1. SPECIATED NMOC COMPOSITION

LANDFILL ID	O	P	Q	R	S	T	U	V	W	X	Y	Z	AA	BB
CHEMICAL NAME														
ACETONE	12		20	1		0	5.33	8.5			32	14	NM	
2 - BUTANOL														
OCTANE														
PENTANE	3.25		0.39	0		0	46.53	0.5			0	45	0	
HEXANE	6.5		6.34	0		13.4	7.13	0			0	25	0	
METHYLESTER ACETIC ACID														
1 - METHOXY - 2 - METHYL PROPANE														
2 - BUTANONE														
1,1 - DICHLOROETHANE	19.5		11.87	2.6	0.053	1.21	6.33	0.45	10		0	7.9	0.1	
1 - BUTANOL														
BUTANE	16.5		0	0		0	6.07	1.5			0	32	0	
4 - METHYL - 2 - PENTANONE														
2 - METHYL PROPANE														
1 - METHYLETHYLESTER BUTANOIC ACID														
2 - METHYL, METHYLESTER PROPANOIC ACID														
CARBON TETRACHLORIDE	0	0.0134	0	0	0	0	0	0	0.0001	0.009	0	0	0	
CHLOROETHANE	1.35		2	4.9	0.026	0.76	7.33	0			0.5	3.7	0	
1,1,3 TRIMETHYL CYCLOHEXANE														
2 - METHYL - 1 - PROPANOL														
1,2 - DICHLOROETHANE	0.45		0	0	0	0	0	0	10	0.176	0	0.1	0	
TRICHLOROFLUOROMETHANE	2.85	0	0.06	2.1	0	0.77	0.5	0.45	0	0	0.2	1.1	0	0
CHLOROMETHANE	0.6		0.7	1.4	0.21	7.19	1.33	1.2			0	3.6	0	
2,5 DIMETHYL FURAN														
2 - METHYL FURAN														
CHLORODIFLUOROMETHANE	0		0	0		0	0	1.9			0	0	0.1	
PROPENE														

TABLE C-1. SPECIATED NMOC COMPOSITION

LANDFILL ID / CHEMICAL NAME	CC	DD	EE	FF	GG	HH	II	JJ	KK	LL	MM	NN	OO	PP
ACETONE		6.5	0	8		19.33			7					7.5
2 - BUTANOL														
OCTANE														
PENTANE		0	1	1.1		0			0					18
HEXANE		0	1.5	3.83		0			1					7
METHYLESTER ACETIC ACID														
1 - METHOXY - 2 - METHYL PROPANE														
2 - BUTANONE														
1,1 - DICHLOROETHANE		2.75	0	0.1	2.3	0.4	0.31		0	1.1	0		2.4	4.5
1 - BUTANOL														
BUTANE		0	2.5	1.13		0			0					9.5
4 - METHYL - 2 - PENTANONE														
2 - METHYL PROPANE														
1 - METHYLETHYLESTER BUTANOIC ACID														
2 - METHYL, METHYLESTER PROPANOIC ACID														
CARBON TETRACHLORIDE		0	0	0		0		68.3	0	0		0.00051		0
CHLOROETHANE		1.45	0.6	4.43	3.65	0			0	0.47				0.85
1,1,3 TRIMETHYL CYCLOHEXANE														
2 - METHYL - 1 - PROPANOL														
1,2 - DICHLOROETHANE		0	0.5	0	1.8				1.9	0.14	0.7	0.122		0
TRICHLOROFLUOROMETHANE	0	3.25	1.05	0.1	0.6	0.67	0	0	0	0.96	0.8	0	0.7	11.9
CHLOROMETHANE		0	0.2	0	0.9	0			0	0.09			2.2	0.24
2,5 DIMETHYL FURAN														
2 - METHYL FURAN														
CHLORODIFLUOROMETHANE		0	1.2	0		0			1		4.8			0.25
PROPENE														

TABLE C-1. SPECIATED NMOC COMPOSITION

LANDFILL ID / CHEMICAL NAME	QQ	RR	SS	TT
ACETONE			0	0
2 - BUTANOL				
OCTANE				
PENTANE			3.96	0.67
HEXANE			6.06	17.96
METHYLESTER ACETIC ACID				
1 - METHOXY - 2 - METHYL PROPANE				
2 - BUTANONE				
1,1 - DICHLOROETHANE			0.71	8.95
1 - BUTANOL				
BUTANE			0	0
4 - METHYL - 2 - PENTANONE				
2 - METHYL PROPANE				
1 - METHYLETHYLESTER BUTANOIC ACID				
2 - METHYL, METHYLESTER PROPANOIC ACID				
CARBON TETRACHLORIDE	0.00063	0.0007	0	0
CHLOROETHANE			0.11	0.95
1,1,3 TRIMETHYL CYCLOHEXANE				
2 - METHYL - 1 - PROPANOL				
1,2 - DICHLOROETHANE	0.056	0.1635	0	0.18
TRICHLOROFLUOROMETHANE	0	0	0.47	0.63
CHLOROMETHANE			1.34	10.22
2,5 DIMETHYL FURAN				
2 - METHYL FURAN				
CHLORODIFLUOROMETHANE			1.33	4.79
PROPENE				

TABLE C-1. SPECIATED NMOC COMPOSITION

LANDFILL ID	A	B	C	D	E	F	G	H	I	J	K	L	M	N
CHEMICAL NAME														
METHYL ISOBUTYL KETONE	0				0		0	2.5		0	0	0.45	0	0.5
ETHYL MERCAPTAN														
DICHLOROFLUOROMETHANE	0.36				5.01		NM	0		0	0	NM	NM	NM
1,1,1 - TRICHLOROETHANE	0.03			5.5	0.48	0.193	0.6	0.37		0.2	0.6	0.03	1.35	0
TETRAHYDROFURAN		30												
ETHYLESTER PROPANOIC ACID		26												
BROMODICHLOROMETHANE	0.22				0.12		0	0		0	0	0	0	0
ETHYL ACETATE														
3 - METHYLHEXANE		20												
C10H16 UNSATURATED HYDROCARBON														
METHYLPROPANE														
CHLOROBENZENE	0.15				0		0	0		0	0	0.05	0	0.2
ACRYLONITRILE	0				0.8		0	0		0	0	0	0	0
METHYLETHYLPROPANOATE														
1,1 - DICHLOROETHENE	0.08				0.23		0.43	0.18	3.1	0.15	0	0	0.1	0.05
METHYL MERCAPTAN														
1,2 - DICHLOROPROPANE	0.06				0.02		0	0		0	0	0	0	0
i - PROPYL MERCAPTAN														
CHLOROFORM	1.56			0.94	0	0.049	0	0		0	0	0	0	0
1,1,2,2 - TETRACHLOROETHANE	0				0		0	0		0	0	0.01	0	0
1,1,2,2 - TETRACHLOROETHENE														
2 - CHLOROETHYLVINYL ETHER	0				0		0	0		0	0	2.25	0	0
t - BUTYL MERCAPTAN														
DIMETHYL SULFIDE														

TABLE C-1. SPECIATED NMOC COMPOSITION

LANDFILL ID	O	P	Q	R	S	T	U	V	W	X	Y	Z	AA	BB
CHEMICAL NAME														
METHYL ISOBUTYL KETONE	1.15		5	1		NM	1	0			11.5	1.2	NM	
ETHYL MERCAPTAN					11									
DICHLOROFLUOROMETHANE	NM		0	NM		NM	NM	NM			NM	NM	NM	
1,1,1 - TRICHLOROETHANE	4.2		0.5	1.3	0	1.24	0.47	0	0.00024	9	0	1.9	0	
TETRAHYDROFURAN														
ETHYLESTER PROPANOIC ACID														
BROMODICHLOROMETHANE	0		2.48	0	0	7.85	0	0	0.001		0	0	0	
ETHYL ACETATE														
3 - METHYLHEXANE														
C10H16 UNSATURATED HYDROCARBON														
METHYLPROPANE														
CHLOROBENZENE	0		10	0	0	0	0	0			0	0	0	
ACRYLONITRILE	0		0	0		7.4	0	0			0	0	0	
METHYLETHYLPROPANOATE														
1,1 - DICHLOROETHENE	0.65		0.75	0	0.04	0	0.13	0			0	0.2	0	0.07
METHYL MERCAPTAN					3.3									
1,2 - DICHLOROPROPANE	1.8		0.5	0	0	0	0.27	0			0	0	0	
i - PROPYL MERCAPTAN					2.1									
CHLOROFORM	0		0	0	0	0	0	0	0.001	0.234	0	0	0	
1,1,2,2 - TETRACHLOROETHANE	0		0	0	0	0	0	0			2.35	0.2	0	
1,1,2,2 - TETRACHLOROETHENE					0.05									
2 - CHLOROETHYLVINYL ETHER	0		0	0	0	0	0	0			0	0	0	
t - BUTYL MERCAPTAN					0.28									
DIMETHYL SULFIDE					0.1									

TABLE C-1. SPECIATED NMOC COMPOSITION

LANDFILL ID	CC	DD	EE	FF	GG	HH	II	JJ	KK	LL	MM	NN	OO	PP
CHEMICAL NAME														
METHYL ISOBUTYL KETONE		4	NM	3.33		3.33			1					0
ETHYL MERCAPTAN										1	23.8			
DICHLOROFLUOROMETHANE		NM	0	NM	NM	NM			NM		1.7			NM
1,1,1 - TRICHLOROETHANE		0.4	0	0	0.25	0	0.016		0	0.37		0.019	0.7	1.15
TETRAHYDROFURAN														
ETHYLESTER PROPANOIC ACID														
BROMODICHLOROMETHANE		0	0	0		0			0	0				0
ETHYL ACETATE													20	
3 - METHYLHEXANE														
C10H16 UNSATURATED HYDROCARBON													15	
METHYLPROPANE													12	
CHLOROBENZENE		0	0	0		0.1			0	0	0.1			0
ACRYLONITRILE		0	0	0		0			0					0
METHYLETHYLPROPANOATE													7.3	
1,1 - DICHLOROETHENE		0.2	0	0	0.1	0			0	0.064	0			0.2
METHYL MERCAPTAN										1	1.3			
1,2 - DICHLOROPROPANE		0.35	0	0		0			0	0.03				0
i - PROPYL MERCAPTAN										1				
CHLOROFORM		0	0	0		0			0	0	0	0.0016		0
1,1,2,2 - TETRACHLOROETHANE		0	0	0		0			0	0				0
1,1,2,2 - TETRACHLOROETHENE										2.6				
2 - CHLOROETHYLVINYL ETHER		0	0	0		0			0	0				0
t - BUTYL MERCAPTAN										1				
DIMETHYL SULFIDE										1				
DICHLOROTETRAFLUOROETHANE											1.1			
DIMETHYL DISULFIDE										1				

TABLE C-1. SPECIATED NMOC COMPOSITION

LANDFILL ID / CHEMICAL NAME	QQ	RR	SS	TT
METHYL ISOBUTYL KETONE				
ETHYL MERCAPTAN				
DICHLOROFLUOROMETHANE			0.48	26.11
1,1,1 - TRICHLOROETHANE	0.0152	0.023	0.16	0.77
TETRAHYDROFURAN				
ETHYLESTER PROPANOIC ACID				
BROMODICHLOROMETHANE			2.02	7.8
ETHYL ACETATE				
3 - METHYLHEXANE				
C10H16 UNSATURATED HYDROCARBON				
METHYLPROPANE				
CHLOROBENZENE			0.43	0
ACRYLONITRILE			0	0
METHYLETHYLPROPANOATE				
1,1 - DICHLOROETHENE			0	0.49
METHYL MERCAPTAN				
1,2 - DICHLOROPROPANE			0.22	0.12
i - PROPYL MERCAPTAN				
CHLOROFORM	0.00278	0.0058	0	0
1,1,2,2 - TETRACHLOROETHANE			0.11	0
1,1,2,2 - TETRACHLOROETHENE				
2 - CHLOROETHYLVINYL ETHER			0	0
t - BUTYL MERCAPTAN				
DIMETHYL SULFIDE				

TABLE C-1. SPECIATED NMOC COMPOSITION

LANDFILL ID	A	B	C	D	E	F	G	H	I	J	K	L	M
CHEMICAL NAME													
DICHLOROTETRAFLUOROETHANE													
DIMETHYL DISULFIDE													
CARBONYL SULFIDE													
1,1,2-TRICHLORO 1,2,2-TRIFLUOROETHANE													
METHYL ETHYL SULFIDE													
1,1,2 - TRICHLOROETHANE	0				0		0	0		0	0	0	0
1,3 - BROMOCHLOROPROPANE													
1,2 - DIBROMOETHANE													
C-1,3 - DICHLOROPROPENE													
t-1,3 - DICHLOROPROPENE													
ACROLEIN	0				0		0	NM		0	0	0	0
1,4 -DICHLOROBENZENE	0				0		0	0		0	0	0	0
BROMOFORM	0				0		0	0		0	0	0	0
1,3 - DICHLOROPROPANE	0				0		0	0		0	0	0	0
1,2 - DICHLOROBENZENE	0				0		0	0		0	0	0	0
1,3 - DICHLORBENZENE	0				0		0	0		0	0	0	0
DIBROMOCHLOROMETHANE	0				0		0	0		0	0	0	0
BROMOMETHANE	0				0		0	0		0	0	0	0

TABLE C-1. SPECIATED NMOC COMPOSITION

LANDFILL ID	N	O	P	Q	R	S	T	U	V	W	X	Y	Z	AA
CHEMICAL NAME														
DICHLOROTETRAFLUOROETHANE														
DIMETHYL DISULFIDE						0.1								
CARBONYL SULFIDE														
1,1,2-TRICHLORO 1,2,2-TRIFLUOROETHANE														
METHYL ETHYL SULFIDE						0.32								
1,1,2 - TRICHLOROETHANE	0	0.1		0	0	0	0	0	0			0	0	0
1,3 - BROMOCHLOROPROPANE										0.005				
1,2 - DIBROMOETHANE										0.0005				
C-1,3 - DICHLOROPROPENE						0								
t-1,3 - DICHLOROPROPENE						0								
ACROLEIN	0	0		NM	0		NM	0	0			0	0	NM
1,4 -DICHLOROBENZENE	0	0		0	0	0	0	0	0			0	0	0
BROMOFORM	0	0		0	0	0	0	0	0			0	0	0
1,3 - DICHLOROPROPANE	0	0		0	0		0	0	0			0	0	0
1,2 - DICHLOROBENZENE	0	0		0	0	0	0	0	0			0	0	0
1,3 - DICHLORBENZENE	0	0		0	0	0	0	0	0			0	0	0
DIBROMOCHLOROMETHANE	0	0		0	0	0	0	0	0			0	0	0
BROMOMETHANE	0	0		0	0	0	0	0	0			0	0	0

TABLE C-1. SPECIATED NMOC COMPOSITION

LANDFILL ID	BB	CC	DD	EE	FF	GG	HH	II	JJ	KK	LL	MM	NN	OO
CHEMICAL NAME														
DICHLOROTETRAFLUOROETHANE												1.1		
DIMETHYL DISULFIDE											1			
CARBONYL SULFIDE											1			
1,1,2-TRICHLORO 1,2,2-TRIFLUOROETHANE												0.5		
METHYL ETHYL SULFIDE														
1,1,2 - TRICHLOROETHANE			0	0	0		0			0	0			
1,3 - BROMOCHLOROPROPANE														
1,2 - DIBROMOETHANE							0							
C-1,3 - DICHLOROPROPENE											0			
t-1,3 - DICHLOROPROPENE											0			
ACROLEIN			0	NM	0		0			0				
1,4 -DICHLOROBENZENE			0	0	0		0			0	0			
BROMOFORM			0	0	0		0			0	0			
1,3 - DICHLOROPROPANE			0	0	0		0			0				
1,2 - DICHLOROBENZENE			0	0	0		0			0	0	0		
1,3 - DICHLORBENZENE			0	0	0		0			0	0	0		
DIBROMOCHLOROMETHANE			0	0	0		0			0	0			
BROMOMETHANE			0	0	0		0			0	0			

TABLE C-1. SPECIATED NMOC COMPOSITION

LANDFILL ID	QQ	RR	SS	TT
CHEMICAL NAME				
DICHLOROTETRAFLUOROETHANE				
DIMETHYL DISULFIDE				
CARBONYL SULFIDE				
1,1,2-TRICHLORO 1,2,2-TRIFLUOROETHANE				
METHYL ETHYL SULFIDE				
1,1,2 - TRICHLOROETHANE			0	0
1,3 - BROMOCHLOROPROPANE				
1,2 - DIBROMOETHANE				
C-1,3 - DICHLOROPROPENE				
t-1,3 - DICHLOROPROPENE				
ACROLEIN			NM	NM
1,4 -DICHLOROBENZENE			0	0
BROMOFORM			0	0
1,3 - DICHLOROPROPANE			0	0
1,2 - DICHLOROBENZENE			0	0
1,3 - DICHLORBENZENE			0	0
DIBROMOCHLOROMETHANE			0	0
BROMOMETHANE			0	0

APPENDIX D: GAS GENERATION RATE MODELING

This appendix provides samples calculations for estimating the landfill air emission rate using the Scholl Canyon model, as well as, a brief discussion of alternative methods. Section D.1 contains a short description of the Scholl Canyon model and sample calculations for 4 model cases. Section D.2 discusses the emission factor method, the SCAQMD method and the Municipal Waste Generation Rate method as alternative techniques for estimating nationwide landfill air emissions.

D.1 Scholl Canyon Model.

The Scholl Canyon model is a single stage, first order kinetic model. It assumes that after a negligible lag time during which anaerobic conditions are established, the gas production rate is at its peak. After the lag time, the gas production rate is assumed to decrease exponentially as the organic fraction of the landfill refuse decreases. The model equation is as follows:[1]

$$\frac{dG}{dt} = kL = kL_0 e^{-kt}$$

where,

$\frac{dG}{dt}$ = methane production rate, ft^3/lb of refuse-yr.

k = rate constant, 1/year

t = time, year

L_0 = total volume of methane ultimately to be produced, ft^3/lb of refuse

If the refuse mass is broken down into the submasses which are placed during each year of the landfill's operation, the model equation is:

$$\frac{dG}{dt} = kL = kL_0 \sum_{i=1}^{n} r_i \exp(-k_i t_i)$$

where,

r_i = fraction of total refuse mass contained in submass i

t_i = time from placement of submass i to point in time at which composite production rate is desired, yr

k_i = gas production rate constant for submass i, 1/year

The rate constant , k, can be calculated if the time and quantity of each refuse submass placement, and the gas flowrate at a given time are known. Once k is calculated from the equation, the methane generation rate at any time can be estimated. Figure D-1 depicts the Scholl Canyon model simulation for two different values of L_0.[2]

D.1.1 Sample Calculations Using Scholl Canyon Model

This section discusses how to use the Scholl Canyon Model to estimate gas generation for several hypothetical landfills (Case 1 through 4 below). In case 1, information on how to estimate the VOC emission rate and toxic compound emission rate is also presented. To use the model, it is necessary for the landfill owner or operator to obtain representative values of gas generation rate, nonmethane organic compound concentration, and toxic compound concentration via field testing (as discussed in Chapter 9.0).

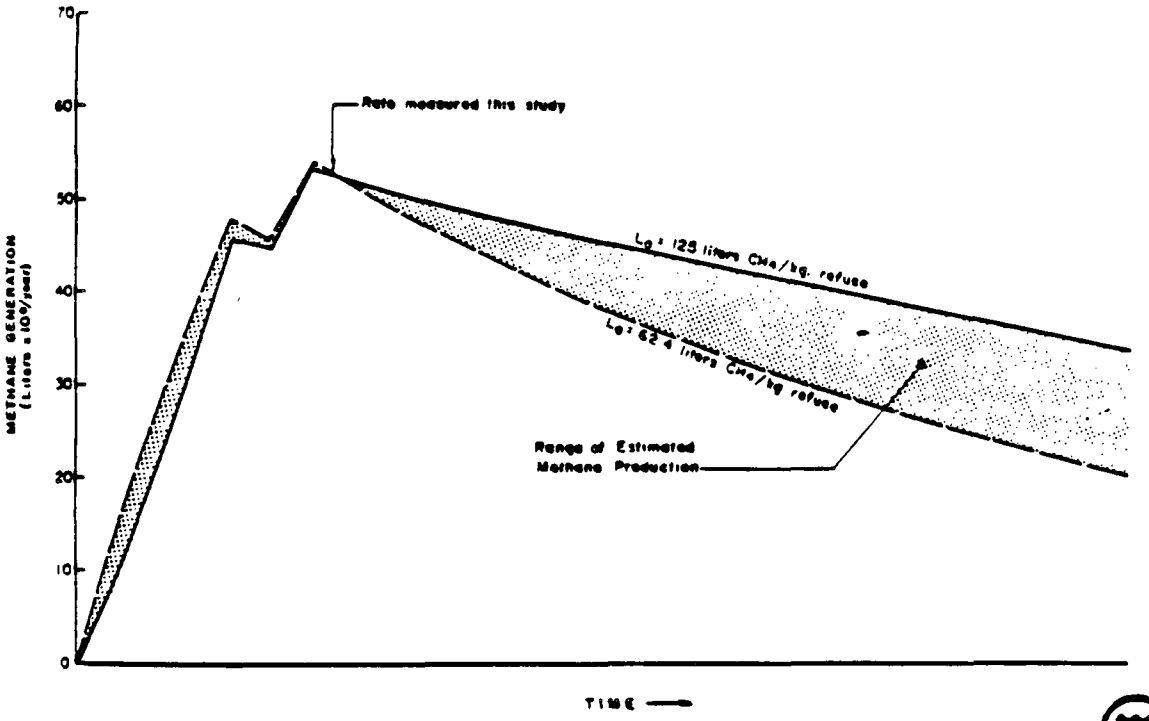

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Figure D-1. Estimated methane production (Scholl Canyon Kinetic Model)

D.1.1.1 Case 1

Given: Landfill A was in operation for 15 years accepting refuse at an average rate of 133,300 Mg/yr. It closed after 15 years of operation with 2×10^6 Mg of refuse in place (RIP). Test well data conducted one year after closure (16 years after initial placement of refuse), indicated that Landfill A is capable of producing 0.0715 ft^3/lb-yr of methane gas. Test well data also showed that the average concentration of nonmethane organic compounds is 1500 ppm and the concentration of toxic compounds is as follows: benzene (120 ppm), methylene chloride (50 ppm), vinyl chloride (100 ppm).

Calculate: Kinetic constant (k), methane generation rate as a function of time, emission rate of VOC, and emission rate of toxic compounds.

1. First, reduce test well data to the actual recoverable methane production rate.

Total recoverable methane gas rate = (test well flowrate)(refuse in place)

$$\text{Total recoverable methane gas rate} = (0.0715)\ (2 \times 10^{12}\ \text{g})\ \frac{\text{lb}}{454\ \text{g}}$$

$$= 315 \times 10^6\ \text{ft}^3\ \text{methane/yr.}$$

2. Calculate the fraction of submass i, r_i, by treating yearly accumulation as the mass of submass i.

$$r_i = \frac{133,300}{2 \times 10^6} = 0.0667$$

3. Calculate the kinetic constant, k, using the recoverable methane gas rate calculated in Step 1 and t of 16 years.

$$\left.\frac{dG}{dt}\right|_{t = 16} = k\, L_o\, M_t \sum_{t_i = 1}^{t_i = 15} \exp\left[-k\,(t_i + t_c)\right]$$

where, t_c = time after closure (= 1 year)

M_t = amount of refuse accumulated at time t

$t_i + t_c$ = age of submass i

[Note that the actual age of the submass i is corrected by adding the time after closure.]

Assuming L_o of 100 liter CH_4/Kg refuse or 3.53×10^3 ft^3 CH_4/Mg refuse,

$$315 \times 10^6 \frac{ft^3\ CH_4}{yr} = k\left(3.53 \times 10^3 \frac{ft^3\ CH_4}{Mg\ refuse}\right)(2 \times 10^6\ Mg\ refuse)$$

$$X \sum_{i = 1}^{i = 15} (0.0667) \exp\left[-k\,(t + 1)\right]$$

$$0.669 = k \sum_{i = 1}^{i = 15} \exp\left[-k\,(t_i + 1)\right]$$

$$= k\,\{\exp(-2k) + \exp(-3k) + \ldots \exp(-16k)\}$$

Solving for k by trial and error procedures, $k \cong 0.1$ 1/yr.

4. Express the model equation with calculated k.

$$\frac{dG}{dt} = k\, L_o\, M_t \sum_{i=1}^{i=15} r_i \exp\left[-k\,(t_i + t_c)\right]$$

$$= (0.1)\,(3.53 \times 10^3)\,(2 \times 10^6)\,(0.0667) \sum_{t_i=1}^{t_i=15} \exp\left[-0.1\,(t_i + t_c)\right]$$

$$= 4.707 \times 10^7 \sum_{t_i=1}^{t_i=15} \exp\left[-0.1\,(t_i + t_c)\right] \text{ in } ft^3 \frac{CH_4}{yr} \qquad (1)$$

5. The future methane gas generation rate now can be calculated by changing t_c. For example, the methane gas generation 5 years after closure may be calculated by setting $t_c = 5$ in Equation (1).

6. The methane gas generation rate before closure can be calculated by modifying the equation (1).

$$\frac{dG}{dt}\text{ (before closure)} = (k\, L_o\, M_n) \sum_{t_i=1}^{t_i=n} \frac{\exp(-k\, t_i)}{(n)} \qquad (2)$$

where, M_n = amount of refuse accumulated over n years.

n = number of years since the initial placement of refuse but before closure

$$\frac{dG}{dt}\text{ (before closure)} = \frac{(0.1)(3.53 \times 10^3)}{n} M_n \sum_{t_i=1}^{t_i=n} \exp(-0.1 t_i)$$

Figure D-2 shows the methane generation rate as a function of time for Landfill A.

7. The VOC emission rate can be calculated by inputting the nonmethane organic compound (i.e. VOC) concentration measured during field testing. The example below represents VOC emissions in year 16 of the landfill.

   - The methane generation rate (315×10^6 ft^3/yr) should be multiplied by 2 to calculate total gas generation. This step assumes that landfill gas is 50 percent methane.

     o $315 \times 10^6\ ft^3/yr \times 2 = 630 \times 10^6\ ft^3/yr$

   - Using the calculated nonmethane organic compound concentration of 1500 ppm and assuming an average VOC molecular weight of 80:

     o $$\frac{630 \times 10^6\ ft^3}{yr} \left| \frac{0.0015\ VOC}{} \right| \frac{lb\ mol}{359\ ft^3} \left| \frac{80\ lb}{lb\ mol} \right.$$

     = 210,000 lb VOC per year
     = 95 Mg VOC per year

8. The toxic compound emission rate can be calculated by inputting the concentration of each toxic compound measured during field testing. The example below represents toxic compound concentration in year 16 of the landfill.

   o $$\frac{630 \times 10^6\ ft^3}{yr} \left| \frac{0.00012\ benzene}{} \right| \frac{lb\ mol}{359\ ft^3} \left| \frac{78\ lb}{lb\ mol} \right.$$

   $$= \frac{16{,}400\ lb\ benzene}{yr} = 7{,}400\ \frac{kg}{yr}\ benzene$$

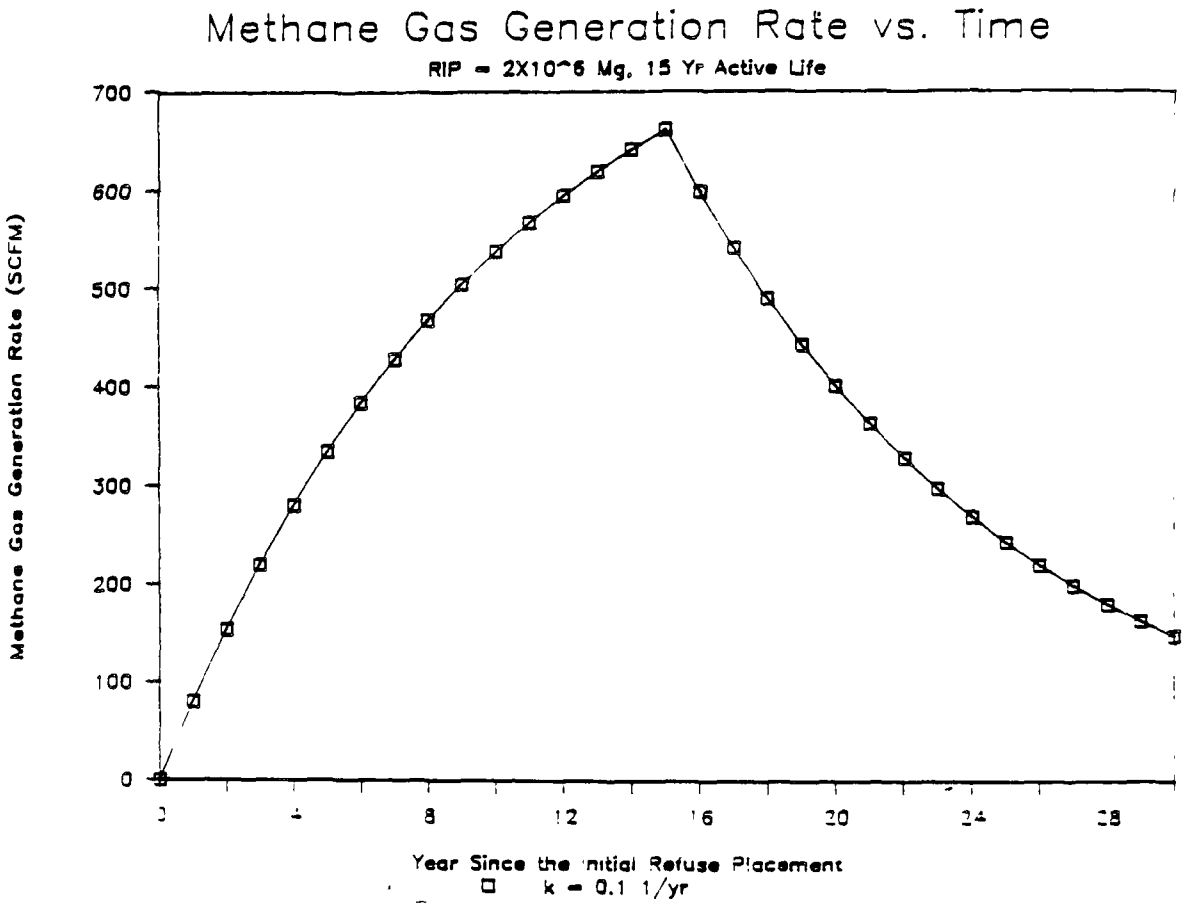

Figure D-2. Methane gas generation rate as a function of time.

o $\frac{630 \times 10^6 \text{ ft}^3}{\text{yr}} \left| \frac{0.00005 \text{ MC}}{} \right| \frac{\text{lb mol}}{359 \text{ ft}^3} \left| \frac{85 \text{ lb}}{\text{lb mol}} \right.$

$= 7{,}450 \frac{\text{lb}}{\text{yr}} \text{ MC} = 3{,}380 \frac{\text{kg}}{\text{yr}} \text{ MC}$

o $\frac{630 \times 10^6 \text{ ft}^3}{\text{yr}} \left| \frac{0.00015 \text{ VC}}{} \right| \frac{\text{lb mol}}{359 \text{ ft}^3} \left| \frac{62 \text{ lb}}{\text{lb mol}} \right.$

$= 16{,}300 \frac{\text{lb}}{\text{yr}}$ Vinyl Chloride $= 7{,}400 \frac{\text{kg}}{\text{yr}}$ VC

D.4.2 Case 2

Given: Landfill B was in operation for 15-years accepting refuse at an average rate of 133,300 Mg/yr. It clsoed after 15 years of operation with 2×10^6 Mg of refuse in place (RIP). Test well data conducted two years after closure (17 years after initial placement of refuse), indicated that Landfill B is capable of producting 0.061 ft^3/lb-yr of methane gas.

Calculate: Kinetic constant (k) and methane generation rate as a function of time.

1. First, reduce test well data to the actual recoverable methane production rate.

Total recoverable methane gas rate = (test well flowrate)(refuse in place)

Total recoverable methane gas rate = $(0.061)\ (2 \times 10^{12}\ g)\ \frac{lb}{454\ g}$

$$= 269 \times 10^6\ ft^3\ \text{methane/yr.}$$

2. Calculate the fraction of submass i, r_i, by treating yearly accumulation as the mass of submass i.

$$r_i = \frac{133,300}{2 \times 10^6} = 0.0667$$

3. Calculate the kinetic constant, k, using the recoverable methane gas rate calculated in Step 1 and t of 17 years.

$$\left.\frac{dG}{dt}\right|_{t = 17} = k\ L_o\ M_t \sum_{t_i = 1}^{t_i = 15} \exp\left[-k\ (t_i + t_c)\right]$$

where, t_c = time after closure (= 2 years)

M_t = amount of refuse accumulated at time t

$t_i + t_c$ = age of submass i

Assuming L_o of 100 liter CH_4/Kg refuse or $3.53 \times 10^3\ ft^3\ CH_4$/Mg refuse,

$$269 \times 10^6\ \frac{ft^3\ CH_4}{yr} = k\ \left(3.53 \times 10^3\ \frac{ft^3\ CH_4}{Mg\ refuse}\right)\ (2 \times 10^6\ Mg\ refuse)$$

$$X \sum_{i = 1}^{i = 15} (0.0667)\ \exp\left[-k\ (t + 2)\right]$$

$$0.571 = k \sum_{i=1}^{i=15} \exp\left[-k\,(t_i + 2)\right]$$

$$= k\,\{\exp(-3k) + \exp(-4k) + \ldots \exp(-17k)\}$$

Solving for k by trial and error procedures, $k \cong 0.2$ 1/yr.

4. Express the model equation with calculated k.

$$\frac{dG}{dt} = k\,L_o\,M_t \sum_{i=1}^{i=15} r_i \exp\left[-k\,(t_i + t_c)\right]$$

$$= (0.2)\,(3.53 \times 10^3)\,(2 \times 10^6)\,(0.0667) \sum_{t_i=1}^{t_i=15} \exp\left[-0.1\,(t_i + t_c)\right]$$

$$= 9.414 \times 10^7 \sum_{t_i=-1}^{t_i=15} \exp\left[-0.2\,(t_i + t_c)\right] \text{ in } ft^3\,\frac{CH_4}{yr}$$

5. The methane gas generation rate before closure can be calculated by:

$$\frac{dG}{dt}\,(\text{before closure}) = (k\,L_o\,M_n) \sum_{t_i=1}^{t_i=n} \frac{\exp(-k\,t_i)}{(n)} \qquad (2)$$

where, M_n = amount of refuse accumulated over n years.

n = number of years since the initial placement of refuse but before closure

$$\frac{dG}{dt}\,(\text{before closure}) = \frac{(0.2)(3.53 \times 10^3)}{n}\,M_n \sum_{t_i=1}^{t_i=n} \exp(-0.2t_i)$$

Figure D-3 shows the methane generation rate as a function of time for Landfill B.

D.4.3 Case 3

Given: Landfill C was in operation for 15 years accepting refuse at an average rate of 333,300 Mg/yr. It closed after 15 years of operiaton with 5×10^6 Mg of refuse in place (RIP). Test well data conducted one year after closure (16 years after initial placement of refuse), indicated that Landfill C is capable of producing 0.0715 ft^3/lb-yr of methane gas.

Calculate: Kinetic constant (k) and methane generation rate as a function of time.

1. First, reduce test well data to the actual recoverable methane production rate.

Total recoverable methane gas rate = (test well flowrate)(refuse in place)

$$\text{Total recoverable methane gas rate} = (0.0715)\ (5 \times 10^{12}\ g)\ \frac{lb}{454\ g}$$

$$= 790 \times 10^6\ ft^3\ \text{methane/yr.}$$

2. Calculate the fraction of submass i, r_i, by treating yearly accumulation as the mass of submass i.

$$r_i = \frac{333,300}{5 \times 10^6} = 0.0667$$

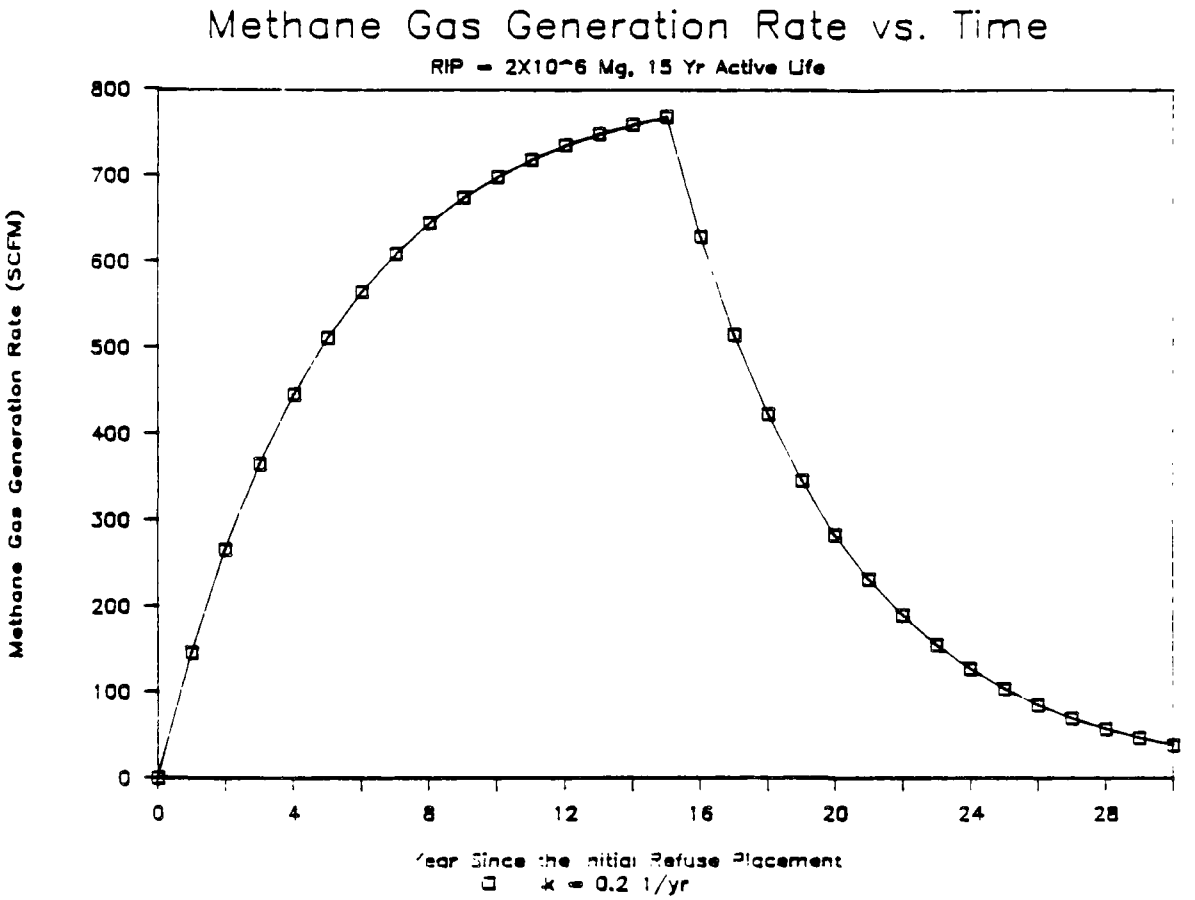

Figure D-3. Methane gas generation rate as a function of time.

3. Calculate the kinetic constant, k, using the recoverable methane gas rate calculated in Step 1 and t of 16 years.

$$\left.\frac{dG}{dt}\right|_{t = 16} = k\, L_o\, M_t \sum_{t_i = 1}^{t_i = 15} \exp\left[-k\,(t_i + t_c)\right]$$

where, t_c = time after closure (= 1 year)

M_t = amount of refuse accumulated at time t

$t_i + t_c$ = age of submass i

Assuming L_o of 100 liter CH_4/Kg refuse or 3.53×10^3 ft^3 CH_4/Mg refuse,

$$790 \times 10^6 \frac{ft^3\ CH_4}{yr} = k\left(3.53 \times 10^3 \frac{ft^3\ CH_4}{Mg\ refuse}\right)(5 \times 10^6\ Mg\ refuse)$$

$$\times \sum_{i = 1}^{i = 15} (0.0667) \exp\left[-k\,(t + 1)\right]$$

$$0.669 = k \sum_{i = 1}^{i = 15} \exp\left[-k\,(t_i + 1)\right]$$

$$= k\,\{\exp(-3k) + \exp(-4k) + \ldots \exp(-17k)\}$$

Solving for k by trial and error procedures, $k \cong 0.1$ 1/yr.

4. Express the model equation with calculated k.

$$\frac{dG}{dt} = k\, L_o\, M_t \sum_{i = 1}^{i = 15} r_i \exp\left[-k\,(t_i + t_c)\right]$$

$$= (0.1)\,(3.53 \times 10^3)\,(5 \times 10^6)\,(0.0667) \sum_{t_i = 1}^{t_i = 15} \exp\left[-0.1\,(t_i + t_c)\right]$$

$$= 11.77 \times 10^7 \sum_{t_i=1}^{t_i=15} \exp\left[-0.1\,(t_i + t_c)\right] \text{ in } ft^3 \frac{CH_4}{yr}$$

5. The methane gas generation rate before closure can be calculated by:

$$\frac{dG}{dt}\text{(before closure)} = (k\,L_o\,M_n) \sum_{t_i=1}^{t_i=n} \frac{\exp(-k\,t_i)}{(n)} \qquad (2)$$

where, M_n = amount of refuse accumulated over n years.

n = number of years since the initial placement of refuse but before closure

$$\frac{dG}{dt}\text{(before closure)} = \frac{(0.1)(3.53 \times 10^3)}{n} M_n \sum_{t_i=1}^{t_i=n} \exp(-0.1t_i)$$

Figure D-4 shows the methane generation rate as a function of time.

D.4.4 <u>Case 4</u>

Given: Landfill D was in operation for 15 years accepting refuse at an average rate of 333,300 Mg/yr. It closed after 15 years of operation with 5×10^6 Mg of refuse in place (RIP). Test well data conducted two years after closure (17 years after initial placement of refuse), indicated that Landfill D is capable of producing 0.061 ft^3/lb-yr of methane gas.

Calculate: Kinetic constant (k) and methane generation rate as a function of time.

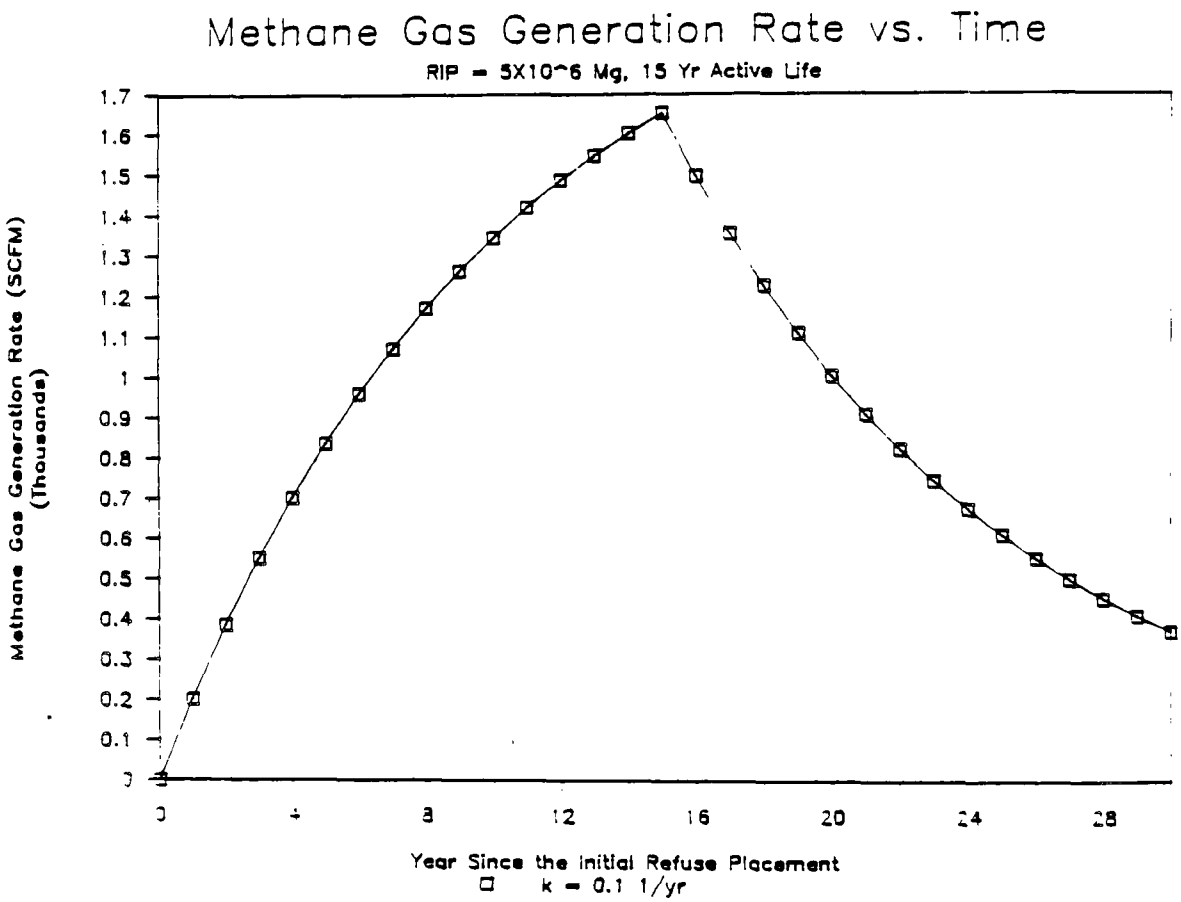

Figure D-4. Methane gas generation rate as a function of time.

1. First, reduce test well data to the actual recoverable methane production rate.

Total recoverable methane gas rate = (test well flowrate)(refuse in place)

$$\text{Total recoverable methane gas rate} = (0.061)\ (5 \times 10^{12}\ \text{g})\ \frac{\text{lb}}{454\ \text{g}}$$

$$= 680 \times 10^{6}\ \text{ft}^{3}\ \text{methane/yr.}$$

2. Calculate the fraction of submass i, r_i, by treating yearly accumulation as the mass of submass i.

$$r_i = \frac{333{,}300}{5 \times 10^{6}} = 0.0667$$

3. Calculate the kinetic constant, k, using the recoverable methane gas rate calculated in Step 1 and t of 17 years.

$$\left.\frac{dG}{dt}\right|_{t = 17} = k\ L_o\ M_t \sum_{t_i = 1}^{t_i = 15} \exp\left[-k\ (t_i + t_c)\right]$$

where, t_c = time after closure (= 2 years)

M_t = amount of refuse accumulated at time t

$t_i + t_c$ = age of submass i

Assuming L_o of 100 liter CH_4/Kg refuse or 3.53×10^{3} ft^3 CH_4/Mg refuse,

$$680 \times 10^{6}\ \frac{\text{ft}^3\ CH_4}{\text{yr}} = k\ \left(3.53 \times 10^{3}\ \frac{\text{ft}^3\ CH_4}{\text{Mg refuse}}\right)\ (5 \times 10^{6}\ \text{Mg refuse})$$

$$X \sum_{i=1}^{i=15} (0.0667) \exp[-k(t+2)]$$

$$0.571 = k \sum_{i=1}^{i=15} \exp[-k(t_i + 2)]$$

$$= k\{\exp(-3k) + \exp(-4k) + \ldots \exp(-17k)\}$$

Solving for k by trial and error procedures, $k \cong 0.2$ 1/yr.

4. Express the model equation with calculated k.

$$\frac{dG}{dt} = k L_o M_t \sum_{i=1}^{i=15} r_i \exp[-k(t_i + t_c)]$$

$$= (0.2)(3.53 \times 10^3)(5 \times 10^6)(0.0667) \sum_{t_i=1}^{t_i=15} \exp[-0.2(t_i + t_c)]$$

$$= 23.54 \times 10^7 \sum_{t_i=1}^{t_i=15} \exp[-0.2(t_i + t_c)] \text{ in } ft^3 \frac{CH_4}{yr}$$

5. The methane gas generation rate before closure can be calculated by:

$$\frac{dG}{dt} \text{(before closure)} = (k L_o M_n) \sum_{t_i=1}^{t_i=n} \frac{\exp(-k t_i)}{(n)}$$

where, M_n = amount of refuse accumulated over n years.

n = number of years since the initial placement of refuse but before closure

$$\frac{dG}{dt} \text{ (before closure)} = \frac{(0.2)(3.53 \times 10^3)}{n} M_n \sum_{t_i = 1}^{t_i = n} \exp(-0.2t_i)$$

Figure D-5 shows the methane generation rate as a function of time.

D.2 Alternative Methods

The emission factor method, the SCAQMD method, and the Municipal Waste Generation method are examples of alternative techniques for estimating landfill air emissions. A comparison of these methods to the Scholl Canyon method is presented in Table D-1. Section D.2.1 describes the emission factor method, while the SCAQMD method and the Municpal Waste Generation method are described in Sections D.2.2 and D.2.3, respectively.

D.2.1 Emission Factor Method.

The emission factor method, like the Scholl Canyon method uses information from the EPA survey of municipal landfills to predict nationwide emission estimates. The design capacity of each eligible landfill is scaled using the appropriate factor, as discussed in Section 3.3.4, and multiplied by an emission factor based on the location of the landfill. The SCAQMD emission factor 13.6 tons NMOC/million tons of refuse-yr and a 2.6 location factor (accounting for gas generation in wet states) can be used. A "wet" state is defined as a state with an annual precipitation of at least 23 inches. Figure D-6 illustrates the calculation scheme.

D.2.2 SCAQMD Method.

An alternate method of estimating the current nationwide landfill air emission rate is to use the SCAQMD 1984 approach which estimated 300 million metric tons of refuse accumulated over 26 years (1957-1983) for 10 million people in the South Coast Air Basin.

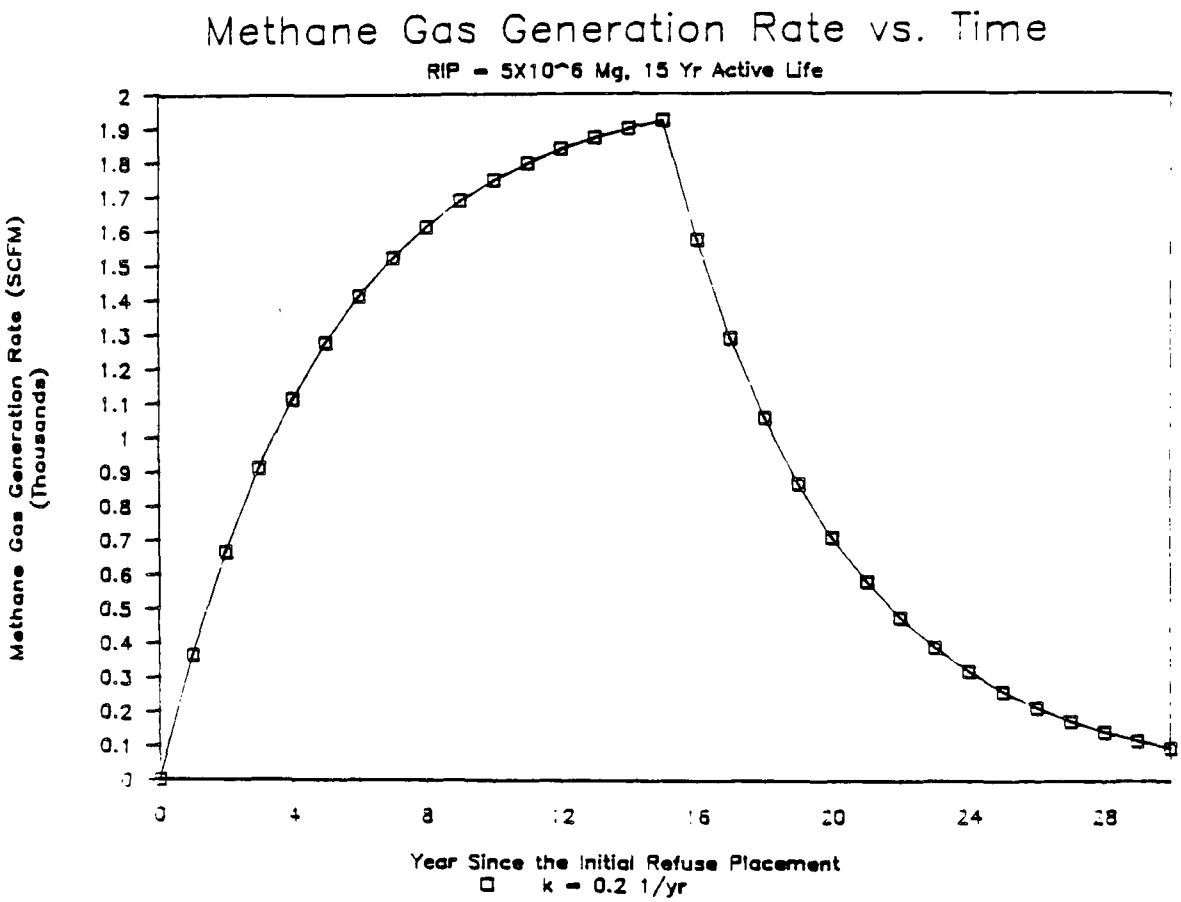

Figure D-5. Methane generation rate as a function of time.

TABLE D-1. NATIONWIDE NMOC EMISSION RATE FROM EXISTING LANDFILLS IN 1987.

Source	Landfill air emission estimation method	Thousand Mg NMOC/yr	Comments
EPA LF Survey	Scholl Canyon	200	Potential NMOC emissions from all existing landfills. Reference year 1992.
EPA LF Survey	Emission Factor	335	Potential NMOC emissions from all existing landfills.
SCAQMD 1984	Based on refuse in place in Southern California generated by 10 million people.	243	"Current" NMOC emissions from all existing active and closed landfills.
1986 EPA-sponsored Study	Based on the yearly estimates of municipal generated from 1960 to 2000.	74.8	"Current" NMOC emissions from all existing active and closed landfills.

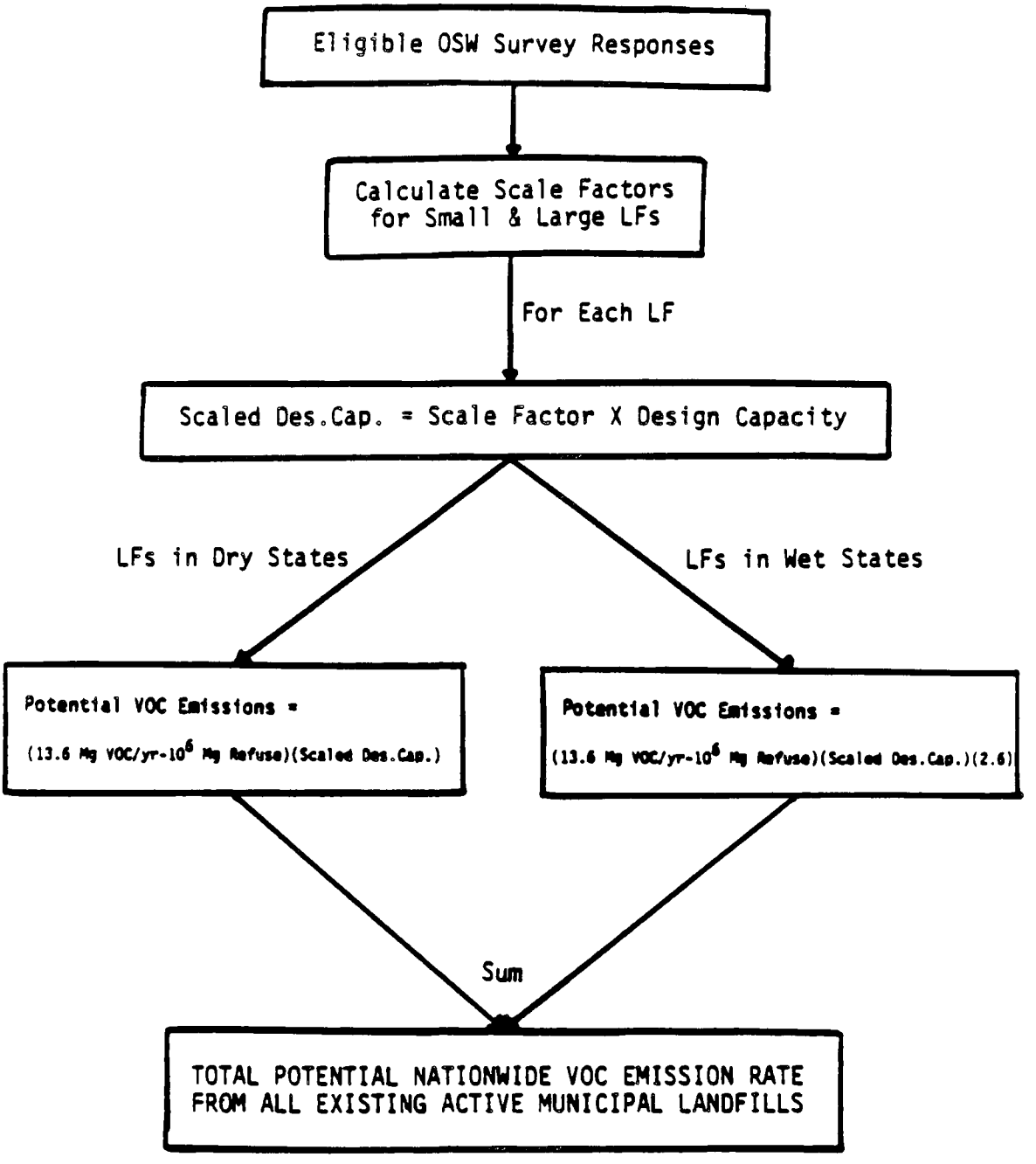

Figure D-6. Calculation schematics for emission factor method.

The in-place refuse for the South Coast in 1983 was estimated using the refuse generation rate per capita and population estimates:

The major assumptions made in the South Coast study were:

- o The average refuse generation rate of 7.9 lbs refuse/capita-day was assumed to be constant over the 26 year period.
- o Refuse has been accumulated since 1957. (Prior to 1957, most of refuse was incinerated).
- o All municipal waste generated is disposed in landfills.

The nationwide landfill air emission rate can be estimated by scaling the SCAQMD refuse in place to the national level. The following additional assumptions were made to scale to the national level:

- o 15 percent of the U.S. population lives in "dry" states and 85% lives in "wet" states.
- o The U.S. population in 1987 is 277 million.
- o The SCAQMD emission factor of 13.6 Mg VOC/million Mg of refuse-yr is used.
- o The emission rate from landfills in "wet" states (>21" of annual precipitation) is 2.6 times greater on a per Mg of refuse basis.

Calculation of the Nationwide landfill air emission rate using this approach is shown below:

- o Current Nationwide VOC Emission Rate from Wet States,

$$= \frac{300 \times 10^6 \text{ Mg refuse}}{10 \times 10^6 \text{ people}} \times 277 \times 10^6 \text{ people} \times 0.85 \times$$

$$\frac{13.6 \text{ Mg VOC}}{\text{yr} - 10^6 \text{ Mg refuse}} \times 2.6 = 249{,}800 \text{ Mg VOC/yr}$$

- o Current Nationwide VOC Emission Rate from Dry States,

$$= \frac{300 \times 10^6 \text{ Mg refuse}}{10 \times 10^6 \text{ people}} \times 277 \times 10^6 \text{ people} \times 0.15 \times$$

$$\frac{13.6 \text{ Mg VOC}}{\text{yr} - 10^6 \text{ Mg refuse}} = 16{,}950 \text{ Mg VOC/yr}$$

- Total Current Nationwide VOC Emission Rate = 267,000 Mg VOC/yr

D.2.3 Municipal Waste Generation Rate Method. The municipal solid waste generation rate from 1960 to 2000 was integrated over the period of 1960 to 1987 (see Figure D-7)[3] to yield the total amount of municipal waste generated over the past 27 years. By assuming that 85 percent of the municipal waste generated is disposed by landfill methods and 85 percent of the U.S.A. population lives in "wet" states, the nationwide landfill air emission rate based on the municipal waste generation rate can be calculated. The assumption that 85 percent of the nationwide municipal waste is based on the estimate provided in an EPA study.[4] The remaining 15 percent is reportedly combusted.

The nationwide landfill air emission rates from new landfills were then calculated using the same calculation scheme shown in Figure D-6. The national potential landfill air emission rate in 1993 and actual landfill air emissionrate expected in 1993 from new landfills are estimated to be 52,000 megagrams/yr and 16,000 megagrams/yr, respectively. The results are also shown in Table D-1.

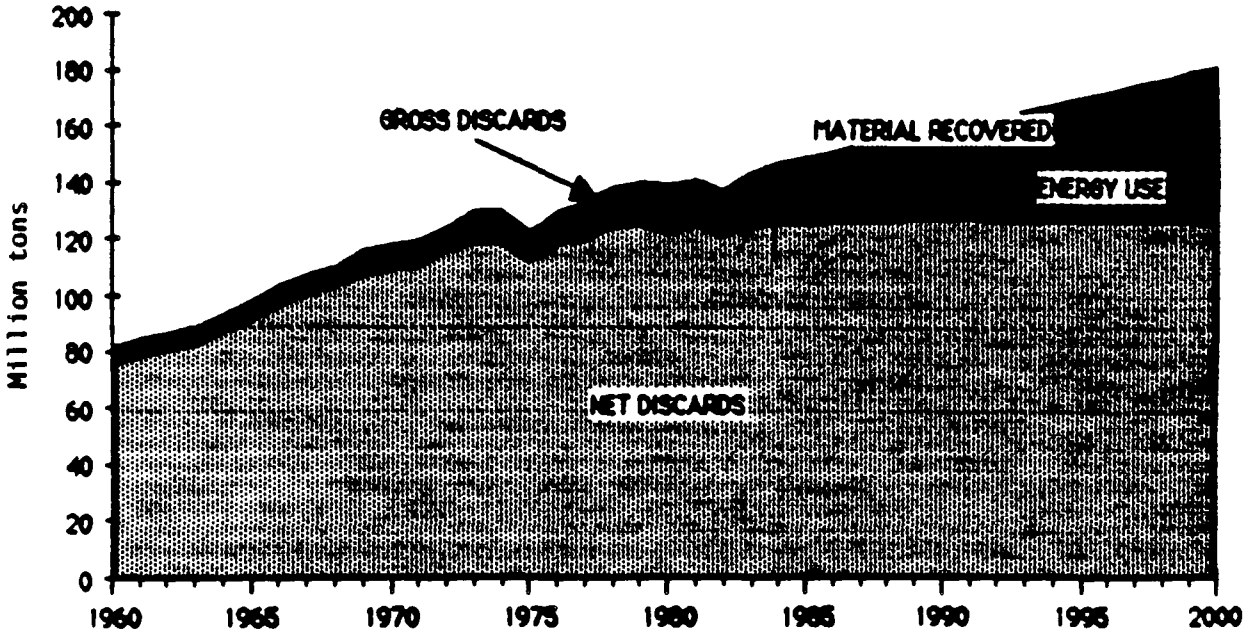

Figure D-7. Gross discards, materials recovery, energy recovery, and discards of municipal solid waste 1960 to 2000.

D.3 REFERENCES

1. Emcon Associates. Methane Generation and Recovery from Landfills. Ann Arbor, Ann Arbor Science. 1982.

2. Reference 1.

3. Franklin Associates, Ltd. Characterization of Municipal Solid Waste in the United States, 1960 to 2000. Final Report. July 11, 1986.

4. The U.S. Environmental Protection Agency. Municpal Waste Combustion Study - Characterization of the Municipal Waste Combustion Industry. EPA/530-SW-87-021h. June 1987.

APPENDIX E

TEST METHODS AND PROCEDURES

APPENDIX E

TEST METHODS AND PROCEDURES

Appendix E contains the three test methods developed by EPA for proposal as part of this rulemaking. These include proposed Method 23 - Determination of Landfill Gas Production Flow Rate, which begins on the following page, proposed Method 3C - Determination of Carbon Dioxide, Methane, Nitrogen, and Oxygen from Stationary Sources, which begins on page E-21, and proposed Method 25C - Determination of Nonmethane Organic Compounds (NMOC) in Landfill Gas, which begins on page E-27.

APPENDIX E - REFERENCE METHODS

METHOD 2E - DETERMINATION OF LANDFILL GAS GAS PRODUCTION FLOW RATE

1. Applicability and Principle

1.1 Applicability. This method applies to the measurement of landfill gas (LFG) production flow rate from municipal solid waste landfills and is used to calculate the flow rate of nonmethane organic compounds (NMOC) from landfills.

1.2 Principle. Extraction wells are installed either in a cluster of three or at five dispersed locations in the landfill. A blower is used to extract LFG from the landfill. LFG composition, landfill pressures, and orifice pressure differentials from the wells are measured and the landfill gas production flow rate is calculated.

1.3 Safety. Since this method is complex, experienced personnel only should perform the test. Explosion-proof equipment shall be used for testing because of the potential explosion hazard of the landfill gas. No smoking shall be allowed on the landfill site during testing. Breathing protection is recommended.

2. Apparatus

2.1 Well Drilling Rig. Capable of boring a 24-in. diameter hole into the landfill to a minimum of 75 percent of the landfill depth. The depth of the well shall not exceed the bottom of the landfill or the liquid level.

2.2 Gravel. No fines, 1 to 3 in. in diameter.

2.3 Bentonite.

2.4 Backfill Material. Clay, soil, and sandy loam have been found to be acceptable.

2.5 Extraction Well Pipe. Polyvinyl chloride (PVC), high density polyethylene (HDPE), fiberglass, or stainless steel, with a minimum diameter of 4 in.

2.6 Well Assembly. PVC ball or butterfly valve, sampling ports at the well head and outlet, and an in-line orifice meter. A schematic of the well assembly is shown in Figure 1.

2.7 Cap. PVC or HDPE.

2.8 Header Piping. PVC or HDPE.

2.9 Auger. Capable of boring a 6- to 9-in. diameter hole to a depth equal to the top of the perforated section of the extraction well, for pressure probe installation.

2.10 Pressure Probe. PVC or stainless steel (316), 1-in. Schedule 40 pipe. Perforate the bottom two thirds. A minimum requirement for perforations is with four 1/4-in. diameter holes spaced 90° apart every 6 in.

2.11 Blower and Flare Assembly. Explosion-proof blower, capable of pulling a vacuum of 25 in. H_2O and of extracting LFG at a flow rate of 300 ft^3/min, a water knockout, and flare or incinerator.

2.12 Standard Pitot Tube and Differential Pressure Gauge for Flow Rate Calibration with Standard Pitot. Same as Method 2, Sections 2.7 and 2.8.

2.13 Orifice Meter. Orifice plate, pressure tabs, and pressure measuring device to measure the LFG flow rate.

2.14 Barometer. Same as Method 4, Section 2.1.5.

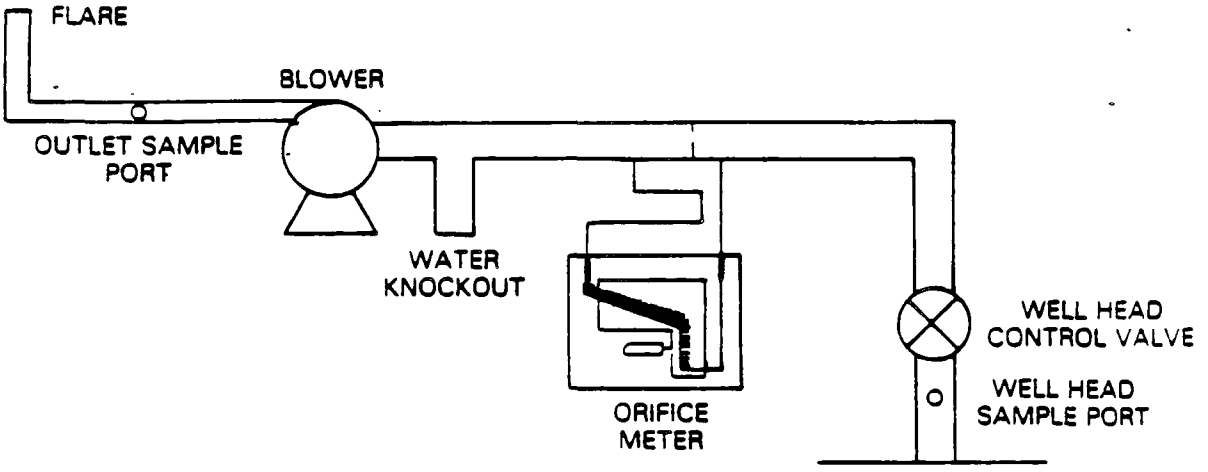

Figure 1. Schematic of above ground assembly.

2.15 Differential Pressure Gauge. Water-filled U-tube manometer or equivalent, capable of measuring within 0.01 in. H_2O, for measuring the pressure of the pressure probes.

3. Procedure

3.1 Placement of Extraction Wells. The landfill owner or operator may install a single cluster of three extraction wells in a test area or space five wells over the landfill. The cluster wells are recommended but may be used only if the composition, age of the refuse, and the landfill depth of the test area can be determined.

3.1.1 Cluster Wells. Consult landfill site records for the age of the refuse, depth, and composition of various sections of the landfill. Select an area near the perimeter of the landfill with a depth equal to or greater than the average depth of the landfill and with the average age of the refuse between 2 and 10 years old. Avoid areas known to contain nondecomposable materials, such as concrete and asbestos. Locate wells as shown in Figure 2.

3.1.1.1 The age of the refuse in a test area will not be uniform, so calculate a weighted average to determine the average age of the refuse as follows.

$$A_{avg} = \sum_{i=1}^{n} f_i A_i$$

where,

A_{avg} = Average age of the refuse tested, yr.

f_i = Fraction of the refuse in the ith section.

A_i = Age of the ith fraction, yr.

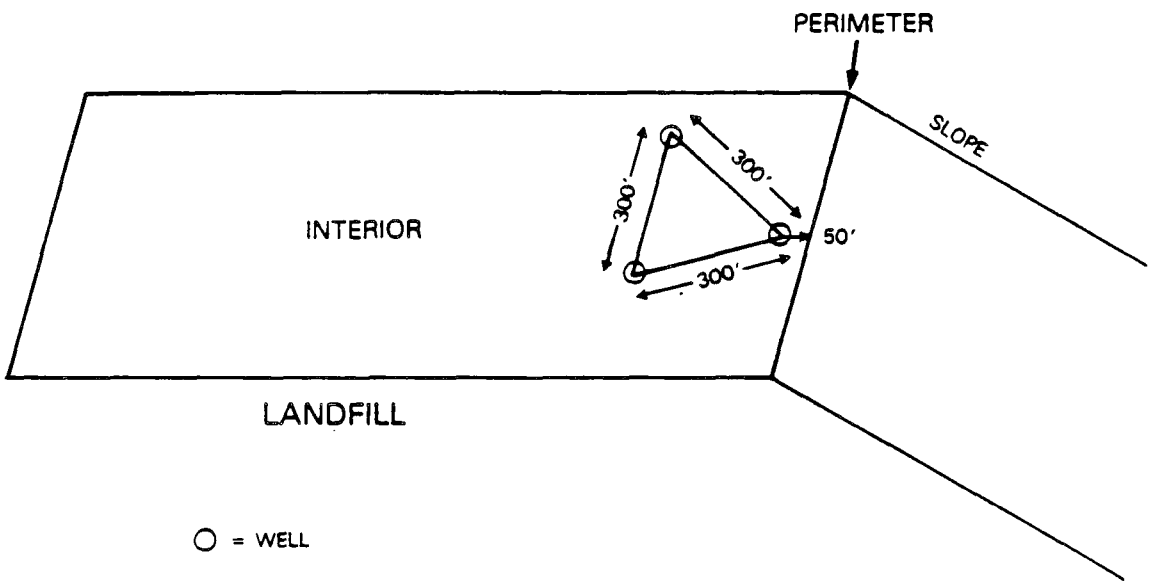

Figure 2. Cluster well placement.

3.1.2 Equal Volume Wells. Divide the sections of the landfill that are at least 2 years old into five areas representing equal volumes. Locate an extraction well near the center of each area.

3.2 Installation of Extraction Wells. Use a well drilling rig to dig a 24-in. diameter hole in the landfill to a minimum of 75 percent of the landfill depth, not to exceed the bottom of the landfill or the liquid level. Perforate the bottom two thirds of the extraction well pipe. A minimum requirement for perforations is with four 1/2-in. diameter holes spaced 90° apart every 4 to 8 in. Place the extraction well in the center of the hole and backfill with gravel to a level 1 ft above the perforated section. Add a layer of backfill material 4 ft thick. Add a layer of bentonite 3 ft thick, and backfill the remainder of the hole with cover material or material equal in permeability to the existing cover material. The specifications for extraction well installation are shown in Figure 3.

3.3 Pressure Probes. Locate pressure probes along three radial arms approximately 120° apart at distances of 10, 50, 100, and 150 ft from the extraction well. The tester has the option of locating additional pressure probes at distances every 50 feet beyond 150 ft. Example placements of probes are shown in Figure 4. The probes 50, 100, and 150 ft (and any additional probes located along the three radial arms) from each well (deep probes) shall extend to a depth equal to the top of the perforated section of the extraction wells. All other probes (shallow probes) shall extend to a depth equal to half the depth of the deep probes.

3.3.1 Use an auger to dig a hole, 6- to 9-in. in diameter, for each pressure probe. Perforate the bottom two thirds of the pressure probe. A minimum requirement for perforations is four 1/4-in. diameter

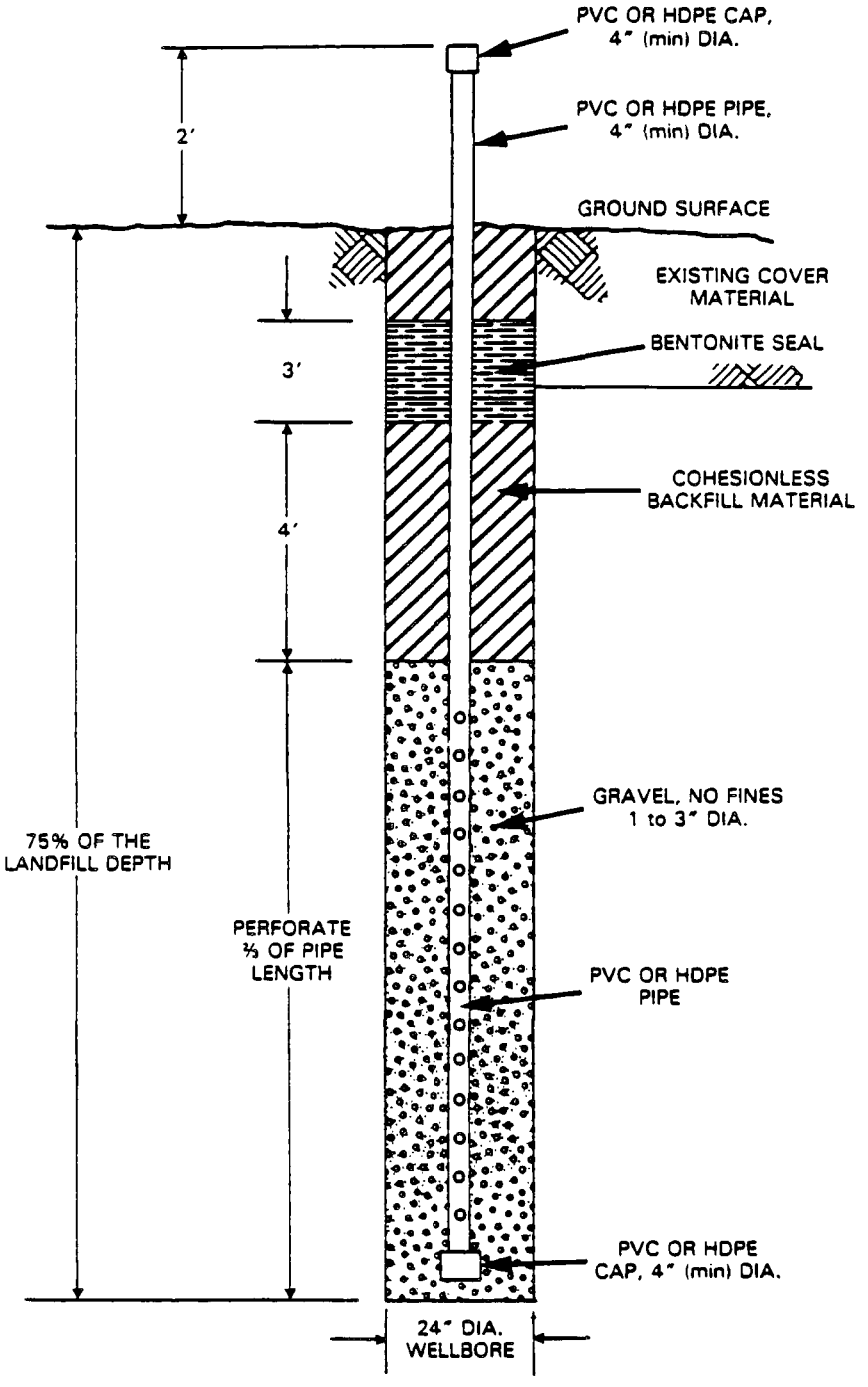

Figure 3. Gas extraction well.

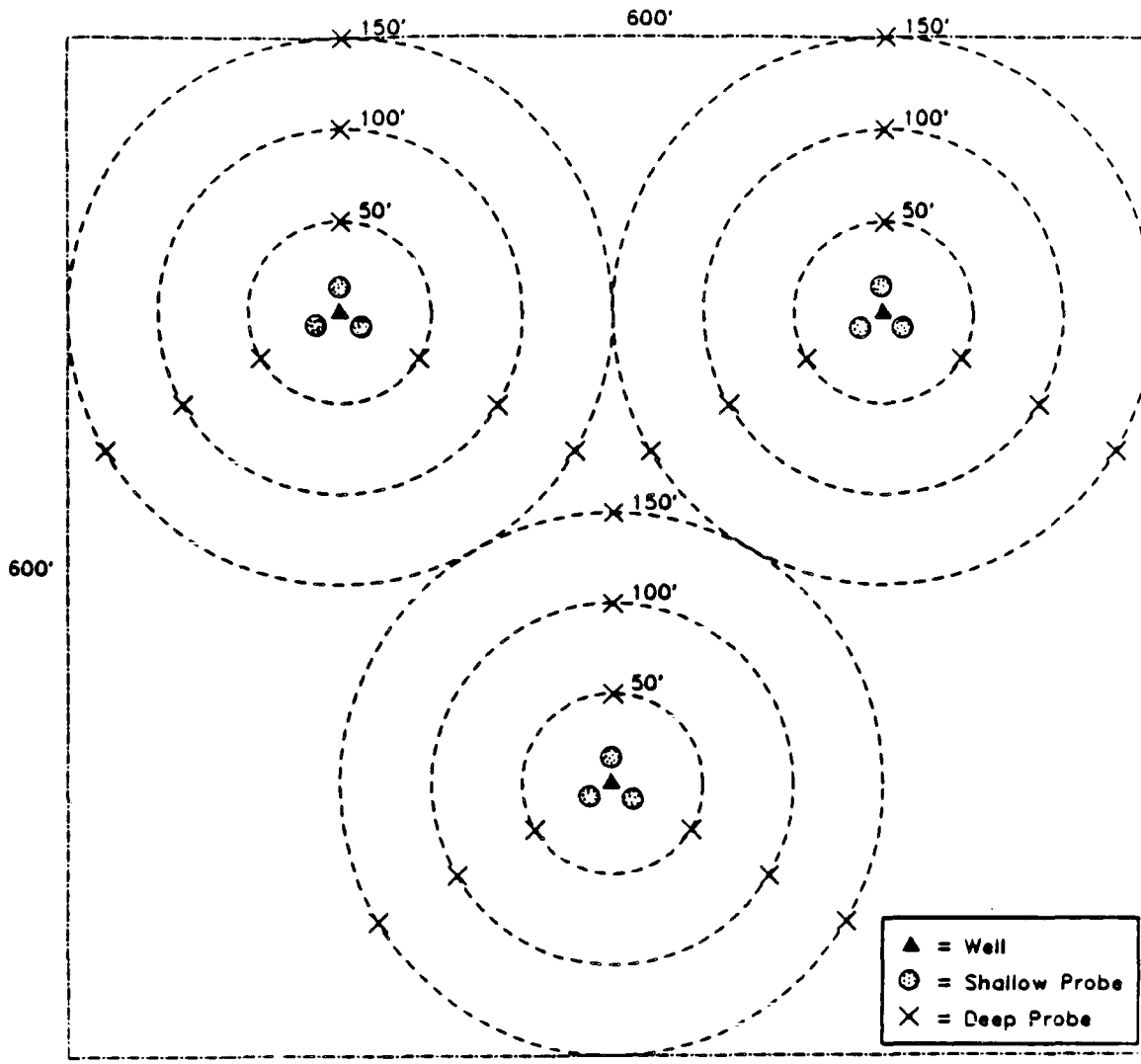

Figure 4. Cluster well configuration.

holes spaced 90^0 apart every 6 in. Place the pressure probe in the center of the hole and backfill with gravel to a level 1 ft above the perforated section. Add a layer of backfill material at least 4 ft thick. Add a layer of bentonite at least 1 ft thick, and backfill the remainder of the hole with cover material or material equal in permeability to the existing cover material. The specifications for pressure probe installation are shown in Figure 5.

3.4 LFG Flow Rate Measurement. Locate an orifice meter as shown in Figure 1. Attach the wells to the blower and flare assembly. The individual wells may be ducted to a common header so that a single blower and flare assembly and orifice meter may be used. Use the procedures in Section 4.1 to calibrate the orifice meter.

3.5 Leak Check. A leak check of the above ground system is required for accurate flow rate measurements and for safety. Sample LFG at the well head sample port and at the outlet sample port. Use Method 3C to determine nitrogen (N_2) concentrations. Determine the difference by using the formula below.

$$\text{Difference} = C_o - C_w$$

where,

C_w = Concentration of N_2 at the wellhead, ppm.

C_o = Concentration of N_2 at the outlet, ppm.

The system passes the leak check if the difference is less than 10,000.

3.6 Static Testing. Close the control valves on the wells during static testing. Measure the gauge pressure (P_g) at each deep pressure probe and the barometric pressure (P_{bar}) every 8 hr for 3 days.

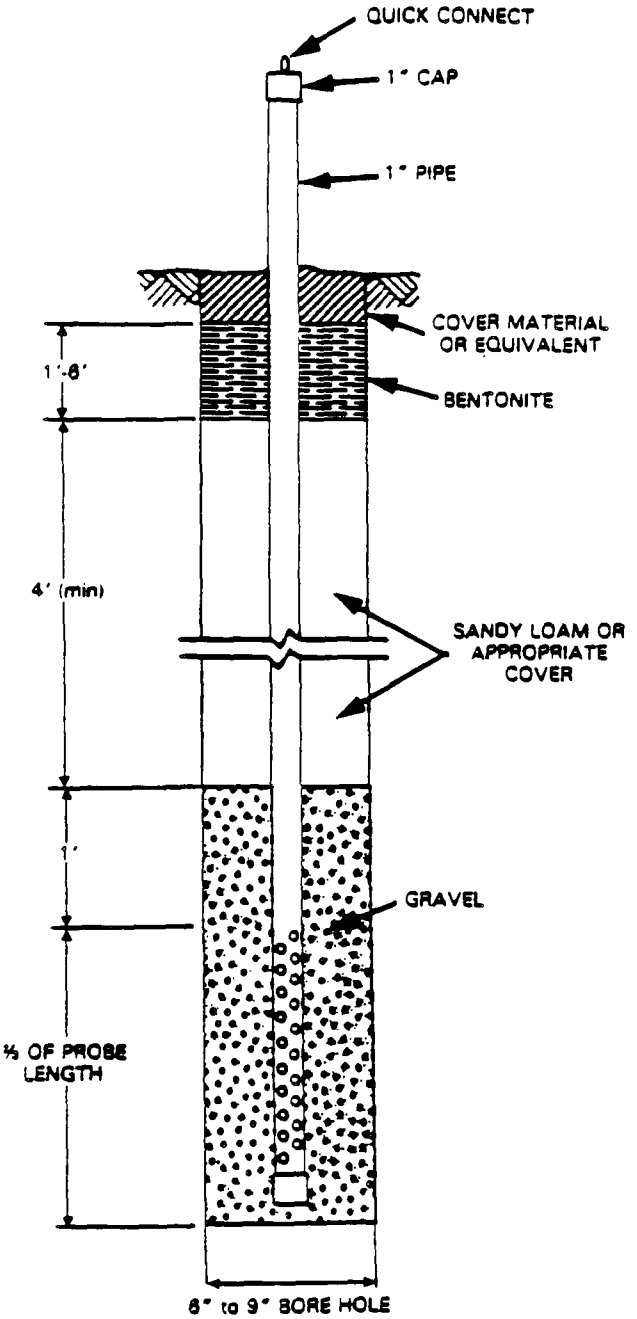

Figure 5. Pressure probe.

Convert the gauge pressure (in. H_2O) of each deep pressure probe to absolute pressure (in. H_2O) by using the following equation. Record as P_i.

$$P_i \text{ (in. } H_2O) = (0.5353)\ P_{bar} \text{ (mm Hg)} + P_g \text{ (in. } H_2O)$$

3.6.1 For each probe, average all of the 8-hr deep pressure probe readings and record as P_{ia}. P_{ia} is used in Section 3.7.6 to determine the maximum radius of influence.

3.6.2 Measure the static flow rate of each well once during static testing.

3.7 Short Term Testing. The purpose of short term testing is to determine the maximum vacuum that can be applied to the wells without infiltration of air into the landfill. The short term testing is done on one well at a time. Burn all LFG with a flare or incinerator.

3.7.1 Use the blower to extract LFG from a single well at twice the static flow rate of the respective well measured in Section 3.6.2. If using a single blower and flare assembly and a common header system, close the control valve on the wells not being measured. Allow 24 hr for the system to stabilize at this flow rate.

3.7.2 Test for infiltration of air into the landfill by measuring the gauge pressures of the shallow pressure probes and using Method 3C to determine the LFG N_2 concentration. If the LFG N_2 concentration is less than 1 percent and all of the shallow probes have a positive gauge pressure, increase the blower vacuum by 2 in. H_2O, wait 24 hr, and repeat the tests for infiltration. Continue the above steps of increasing blower vacuum by 2 in. H_2O, waiting 24 hr, and testing for infiltration until the concentration of N_2 exceeds 1 percent or any of

the shallow probes have a negative gauge pressure, at which time reduce the blower vacuum so that the N_2 concentration is less than 1 percent and the gauge pressures of the shallow probes are positive.

3.7.3 At this blower vacuum, measure P_{bar} every 8 hr for 24 hr and record the LFG flow rate as Q_s and the probe gauge pressures for all of the probes as P_f. Convert the gauge pressures of the deep probes to absolute pressures for each 8 hr reading at Q_s as follows.

$$P_f \text{ (in. } H_2O) = (0.5353)\ P_{bar} \text{ (mm Hg)} + P_f \text{ (in. } H_2O)$$

3.7.4 For each probe, average the 8-hr deep pressure probe readings and record as P_{fa}.

3.7.5 For each probe, compare the initial average pressure (P_{ia}) from Section 3.6.1 to the final average pressure (P_{fa}). Determine the furthermost point from the well head along each radial arm where $P_{fa} \leq P_{ia}$. This distance is the maximum radius of influence, which is the distance from the well affected by the vacuum. Average these values to determine the average maximum radius of influence (R_{ma}).

3.7.7 Calculate the depth (D) affected by the extraction well as follows.

$$D_{st} = WD + R_{ma}$$

where,

WD = Well depth, ft.

3.7.8 Calculate the void volume for the extraction well (V) as follows.

$$V = 0.40 \quad R_{ma}^{2} \, D_{st}$$

3.7.9 Repeat the procedures in Section 3.7 for each well.

3.8 Calculate the total void volume of the test wells (V_v) by summing the void volumes (V) of each well.

3.9 Long Term Testing. The purpose of long term testing is to extract two void volumes of LFG from the extraction wells. Use the blower to extract LFG from the wells. If a single blower and flare assembly and common header system are used, open all control valves and set the blower vacuum equal to the highest stabilized blower vacuum demonstrated by any individual well in Section 3.7. Every 8 hr, sample the LFG from the well head sample port, measure the gauge pressures of the shallow pressure probes, the blower vacuum, the LFG flow rate, and use the criteria for infiltration in Section 3.7.2 and Method 3C to test for infiltration. If Infiltration is detected, do not reduce the blower vacuum, but reduce the LFG flow rate from the well by adjusting the control valve on the well head. Continue until the equivalent of two total void volumes (V_v) have been extracted, or until $V_t = 2 \, V_v$.

3.9.1 Calculate V_t, the total volume of LFG extracted from the wells, as follows.

$$V_t = \sum_{i=1}^{n} 60 \, Q_i \, t_{vi}$$

where,

V_t = Total volume of LFG extracted from wells, ft^3.

Q_i = LFG flow rate measured at orifice meter at the ith interval, ft^3/min.

t_{vi} = Time of the ith interval (usually 8), hr.

3.9.2 Record the final stabilized flow rate as Q_f. If, during the long term testing, the flow rate does not stabilize, calculate Q_f by averaging the last 10 recorded flow rates.

3.9.3 For each deep probe, convert each gauge pressure to absolute pressure as in Section 3.7.4. Average these values and record as P_{sa}. For each probe, compare P_{ia} to P_{sa}. Determine the furthermost point from the well head along each radial arm where $P_{sa} \leq P_{ia}$. This distance is the stabilized radius of influence. Average these values to determine the average stabilized radius of influence (R_{sa}).

3.10 Determine the NMOC mass emission rate using the procedures in Section 5.

4. Calibrations

4.1 Orifice Calibration Procedure. Locate a standard pitot tube in line with an orifice meter. Use the procedures in Section 3 of Method 2 to determine the average dry gas volumetric flow rate for at least five flow rates that bracket the expected LFG flow rates, except in Section 3.1, use a standard pitot tube rather than a Type S pitot tube. Method 3C may be used to determine the dry molecular weight. It may be necessary to calibrate more than one orifice meter in order to bracket the LFG flow rates. Construct a calibration curve by plotting the pressure drops across the orifice meter for each flow rate versus the average dry gas volumetric flow rate in ft^3/min of the gas.

5. Calculations

5.1 Nomenclature.

A_{avg} = Average age of the refuse tested, yr.

A_i = Age of refuse in the ith fraction, yr.

A = Age of landfill, yr.

A_r = Acceptance rate, Mg/yr.

C = NMOC concentration, ppm.

D = Depth affected by the test wells, ft.

D_{st} = Depth affected by the test wells in the short term test, ft.

f = Fraction of decomposable refuse in the landfill.

f_i = Fraction of the refuse in the ith section.

k = Landfill gas generation constant, yr^{-1}.

L_o = Methane generation potential, ft^3/Mg.

L_o' = Revised methane generation potential to account for the amount of nondecomposable material in the landfill, ft^3/Mg.

M_i = Mass of refuse of the ith section, Mg.

M_r = Mass of decomposable refuse affected by the test well, Mg.

P_{bar} = Atmospheric pressure, mm Hg.

P_g = Gauge pressure of the deep pressure probes, in. H_2O.

P_i = Initial absolute pressure of the deep pressure probes during static testing, in. H_2O.

P_{ia} = Average initial absolute pressure of the deep pressure probes during static testing, in. H_2O.

P_f = Final absolute pressure of the deep pressure probes during short term testing, in. H_2O.

P_{fa} = Average final absolute pressure of the deep pressure probes during short term testing, in. H_2O.

P_s = Final absolute pressure of the deep pressure probes during long term testing, in. H_2O.

P_{sa} = Average final absolute pressure of the deep pressure probes during long term testing, in. H_2O.

Q_f = Final stabilized flow rate, ft^3/min.

Q_i = LFG flow rate measured at orifice meter during the ith interval, ft^3/min.

Q_s = Maximum LFG flow rate at each well determined by short term test, ft^3/min.

Q_t = NMOC mass emission rate, ft^3/min.

R_m = Maximum radius of influence, ft.

R_{ma} = Average maximum radius of influence, ft.

R_s = Stabilized radius of influence for an individual well, ft.

R_{sa} = Average stabilized radius of influence, ft.

t_i = Age of section i, yr.

t_t = Total time of long term testing, yr.

V = Void volume of test well, ft^3.

V_r = Volume of refuse affected by the test well, ft^3.

V_t = Total volume of refuse affected by the long term testing, ft^3.

V_v = Total void volume affected by test wells, ft^3.

WD = Well depth, ft.

ρ = refuse density, Mg/ft^3 (Assume 0.018 Mg/ft^3 if data are unavailable).

5.2 Use the following equation to calculate the depth affected by the test well. If using cluster wells, use the average depth of the wells for WD.

$$D = WD + R_{sa}$$

5.3 Use the following equation to calculate the volume of refuse affected by the test well.

$$V_r = R_{sa} \pi D$$

5.4 Use the following equation to calculate the mass affected by the test well.

$$M_r = V_r \rho$$

5.5 Modify L_o to account for the nondecomposable refuse in the landfill.

$$L_o' = f L_o$$

5.6 In the following equation, solve for k by iteration. A suggested procedure is to select a value for k, calculate the left side

of the equation, and if not equal to zero, select another value for k. Continue this process until the left hand side of the equation equals zero, ±0.001.

$$k\ e^{-k\ A_{avg}} - \left[\frac{Q_f}{2\ L_o'\ M_r} \right] = 0$$

5.7 Use the following equation to determine landfill NMOC mass emission rate if the yearly acceptance rate of refuse has been consistent (±10 percent) over the life of the landfill.

$$Q_t = 2\ L_o'\ A_r\ (1 - e^{-k\ A})\ C\ (1.018 \times 10^{-10})$$

5.8 Use the following equation to determine landfill NMOC mass emission rate if the acceptance rate has not been consistent over the life of the landfill.

$$Q_t = 2\ k\ L_o'\ C\ (1.018 \times 10^{-10}) \sum_{i=1}^{n} M_i\ e^{-kt_i}$$

6. Bibliography

1. Same as Method 2, Appendix A, 40 CFR Part 60.

2. Emcon Associates, Methane Generation and Recovery from Landfills. Ann Arbor Science, 1982.

3. The Johns Hopkins University, Brown Station Road Landfill Gas Resource Assessment, Volume 1: Field Testing and Gas Recovery Projections. Laurel, Maryland: October 1982.

4. Mandeville and Associates, Procedure Manual for Landfill Gases Emission Testing.

5. Letter and attachments from Briggum, S., Waste Management of North America, to Thorneloe, S., EPA. Response to July 28, 1988 request for additional information. August 18,1988.

6. Letter and attachments from Briggum, S., Waste Management of North America, to Wyatt, S., EPA. Response to December 7, 1988 request for additional information. January 16, 1989.

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METHOD 3C - DETERMINATION OF CARBON DIOXIDE, METHANE, NITROGEN, AND OXYGEN FROM STATIONARY SOURCES

1. Applicability and Principle

1.1 Applicability. This method applies to the analysis of carbon dioxide (CO_2), methane (CH_4), nitrogen (N_2), and oxygen (O_2) in samples from municipal landfills and other sources when specified in an applicable subpart of the regulations.

1.2 Principle. A portion of the sample is injected into a gas chromatograph (GC) and the CO_2, CH_4, N_2, and O_2 concentrations are determined by using a thermal conductivity detector (TCD) and integrator.

2. Range and Sensitivity

2.1 Range. The range of this method depends upon the concentration of samples. The analytical range of TCD's is generally between approximately 10 ppm and the upper percent range.

2.2 Sensitivity. The sensitivity limit for a compound is defined as the minimum detectable concentration of that compound, or the concentration that produces a signal-to-noise ratio of three to one. For CO_2, CH_4, N_2, and O_2, the sensitivity limit is in the low ppm range.

3. Interferences

Since the TCD exhibits universal response and detects all gas components except the carrier, interferences may occur. Choosing the appropriate GC or shifting the retention times by changing the column flow rate may help to eliminate resolution interferences.

To assure consistent detector response, helium is used to prepare calibration gases. Frequent exposure to samples or carrier gas containing oxygen may gradually destroy filaments.

4. Apparatus

4.1 Gas Chromatograph. GC having at least the following components:

4.1.1 Separation Column. Appropriate column(s) to resolve CO_2, CH_4, N_2, O_2, and other gas components that may be present in the sample. One column that has been advertised to work in this case is column CTR I available from Alltech Associates Inc., 2051 Waukegan Road, Deerfield, Illinois 60015. NOTE: Mention of trade names or specific products does not constitute endorsement or recommendation by the U. S. Environmental Protection Agency.

4.1.2 Sample Loop. Teflon or stainless steel tubing of the appropriate diameter. NOTE: Mention of trade names or specific products does not constitute endorsement or recommendation by the U. S. Environmental Protection Agency.

4.1.3 Conditioning System. To maintain the column and sample loop at constant temperature.

4.1.4 Thermal Conductivity Detector.

4.2 Recorder. Recorder with linear strip chart. Electronic integrator (optional) is recommended.

4.3 Teflon Tubing. Diameter and length determined by connection requirements of cylinder regulators and the GC.

4.4 Regulators. To control gas cylinder pressures and flow rates.

4.5 Adsorption Tubes. Applicable traps to remove any O_2 from the carrier gas.

5. Reagents

5.1 Calibration and Linearity Gases. Standard cylinder gas mixtures for each compound of interest with at least three concentration levels spanning the range of suspected sample concentrations. The calibration gases shall be prepared in helium.

5.2 Carrier Gas. Helium, high-purity.

6. Analysis

6.1 Sample Collection. Use the sample collection procedures described in Methods 3 or 25C to collect a sample of landfill gas (LFG).

6.2 Preparation of GC. Before putting the GC analyzer into routine operation, optimize the operational conditions according to the manufacturer's specifications to provide good resolution and minimum analysis time. Establish the appropriate carrier gas flow and set the detector sample and reference cell flow rates at exactly the same levels. Adjust the column and detector temperatures to the recommended levels. Allow sufficient time for temperature stabilization. This may typically require 1 hour for each change in temperature.

6.3 Analyzer Linearity Check and Calibration. Perform this test before sample analysis. Using the gas mixtures in Section 5.1, verify the detector linearity over the range of suspected sample concentrations with at least three points per compound of interest. This initial check may also serve as the initial instrument calibration. All subsequent calibrations may be performed using a single-point standard gas provided the calibration point is within 20 percent of the sample component concentration. For each instrument calibration, record the carrier and detector flow rates, detector filament and block temperatures, attenuation factor, injection time,

chart speed, sample loop volume, and component concentrations. Plot a linear regression of the standard concentrations versus area values to obtain the response factor of each compound. Alternatively, response factors of uncorrected component concentrations (wet basis) may be generated using instrumental integration. NOTE: Peak height may be used instead of peak area throughout this method.

6.4 Sample Analysis. Purge the sample loop with sample, and allow to come to atmospheric pressure before each injection. Analyze each sample in duplicate, and calculate the average sample area (A). The results are acceptable when the peak areas for two consecutive injections agree within five percent of their average. If they do not agree, run additional samples until consistent area data are obtained. Determine the tank sample concentrations according to Section 7.2.

7. Calculations

Carry out calculations retaining at least one extra decimal figure beyond that of the acquired data. Round off results only after the final calculation.

7.1 Nomenclature.

A = Average sample area.

B_w = Moisture content in the sample, fraction.

C = Component concentration in the sample, dry basis, ppm.

C_t = Calculated NMOC concentration, ppm C equivalent.

C_{tm} = Measured NMOC concentration, ppm C equivalent.

P_{bar} = Barometric pressure, mm Hg.

P_{ti} = Gas sample tank pressure after evacuation, mm Hg absolute.

P_t = Gas sample tank pressure after sampling, but before pressurizing, mm Hg absolute.

P_{tf} = Final gas sample tank pressure after pressurizing, mm Hg absolute.

P_w = Vapor pressure of H_2O (from Table 3C-1), mm Hg.

T_{ti} = Sample tank temperature before sampling, ^oK.

T_t = Sample tank temperature at completion of sampling, ^oK.

T_{tf} = Sample tank temperature after pressurizing, ^oK.

r = Total number of analyzer injections of sample tank during analysis (where j = injection number, 1...r).

R = Mean calibration response factor for specific sample component, area/ppm.

7.2 Concentration of Sample Components. Calculate C for each compound using Equations 3C-1 and 3C-2. Use the temperature and barometric pressure at the sampling site to calculate B_w. If the sample was diluted with helium using the procedures in Method 25C, use Equation 3C-3 to calculate the concentration.

$$B_w = \frac{P_w}{P_{bar}} \qquad \text{3C-1}$$

$$C = \frac{A}{R(1-B_w)} \qquad \text{3C-2}$$

$$C = \left[\frac{\frac{P_{tf}}{T_{tf}}}{\frac{P_t}{T_t} - \frac{P_{ti}}{T_{ti}}}\right] \frac{A}{R(1-B_w)} \qquad \text{3C-3}$$

8. Bibliography

1. McNair, H.M., and E.J. Bonnelli. Basic Gas Chromatography. Consolidated Printers, Berkeley, CA. 1969.

TABLE 3C-1. MOISTURE CORRECTION

Temperature, °C	Vapor pressure of H_2O, mm Hg	Temperature, °C	Vapor pressure of H_2O, mm Hg
4	6.1	18	15.5
6	7.0	20	17.5
8	8.0	22	19.8
10	9.2	24	22.4
12	10.5	26	25.2
14	12.0	28	28.3
16	13.6	30	31.8

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METHOD 25C- DETERMINATION OF NONMETHANE ORGANIC COMPOUNDS (NMOC) IN LANDFILL GASES

1. Applicability and Principle

1.1 Applicability. This method is applicable to the sampling and measurement of nonmethane organic compounds (NMOC) as carbon in landfill gases.

1.2 Principle. A sample probe that has been perforated at one end is driven or augered to a depth of 3 feet (ft) below the bottom of the landfill cover. A sample of the landfill gas is extracted with an evacuated cylinder. The NMOC content of the gas is determined by injecting a portion of the gas into a gas chromatographic column to separate the NMOC from carbon monoxide (CO), carbon dioxide (CO_2), and methane (CH_4); the NMOC are oxidized to CO_2, reduced to CH_4, and measured by a flame ionization detector (FID). In this manner, the variable response of the FID associated with different types of organics is eliminated.

2. Apparatus

2.1 Sample Probe. Stainless steel, with the bottom third perforated. The sample probe shall be capped at the bottom and shall have a threaded cap with a sampling attachment at the top. The sample probe shall be long enough to go through and extend no less than 3 ft below the landfill cover. If the sample probe is to be driven into the landfill, the bottom cap should be designed to facilitate driving the probe into the landfill.

2.2 Sampling Train.

2.2.1 Rotameter with Flow Control Valve. Capable of measuring a sample flow rate of 100 ± 10 ml/min. The control valve shall be made of stainless steel.

2.2.2 Sampling Valve. Stainless steel.

2.2.3 Pressure Gauge. U-tube mercury manometer, or equivalent, capable of measuring pressure to within 1 mm Hg in the range of 0 to 1,100 mm Hg.

2.2.4 Sample Tank. Stainless steel or aluminum cylinder, with a minimum volume of 4 liters and equipped with a stainless steel sample tank valve.

2.3 Vacuum Pump. Capable of evacuating to an absolute pressure of 10 mm Hg.

2.4 Purging Pump. Portable, explosion proof, and suitable for sampling NMOC.

2.5 Pilot Probe Procedure. The following are needed only if the tester chooses to use the procedure described in Section 4.2.1.

2.5.1 Pilot Probe. Tubing of sufficient strength to withstand being driven into the landfill by a post driver and an outside diameter of at least 0.25 in. smaller than the sample probe. The pilot probe shall be capped on both ends and long enough to go through the landfill cover and extend no less than 3 ft into the landfill.

2.5.2 Post Driver and Compressor. Capable of driving the pilot probe and the sampling probe into the landfill. The Kitty Hawk portable post driver has been found to be acceptable. NOTE: Mention of trade names or specific products does not constitute endorsement by the Environmental Protection Agency.

2.6 Auger Procedure. The following are needed only if the tester chooses to use the procedure described in Section 4.2.2.

2.6.1 Auger. Capable of drilling through the landfill cover and to a depth of no less than 3 ft into the landfill.

2.6.2 Pea Gravel.

2.6.3 Bentonite.

2.7 NMOC Analyzer, Barometer, Thermometer, and Syringes. Same as in Sections 2.3, 2.4.1, 2.4.2, 2.4.4, respectively, of Method 25.

3. Reagents

3.1 NMOC Analysis. Same as in Method 25, Section 3.2.

3.2 Calibration. Same as in Method 25, Section 3.4, except omit Section 3.4.3.

4. Procedure

4.1 Sample Tank Evacuation and Leak Check. Conduct the sample tank evacuation and leak check either in the laboratory or the field. Connect the pressure gauge and sampling valve to the sample tank. Evacuate the sample tank to 10 mm Hg absolute pressure or less. Close the sampling valve, and allow the tank to sit for 60 minutes. The tank is acceptable if no change is noted. Include the results of the leak check in the test report.

4.2 Sample Probe Installation. The tester may use the procedure in Sections 4.2.1 or 4.2.2. CAUTION: LFG contains methane and therefore explosive mixtures may exist on or near the landfill. It is advisable to take appropriate safety precautions when testing landfills, such as refraining from smoking.

4.2.1 Pilot Probe Procedure. Use the post driver to drive the pilot probe at least 3 ft below the landfill cover. Alternative

procedures to drive the probe into the landfill may be used subject to the approval of the Administrator.

4.2.1.1 Remove the pilot probe and drive the sample probe into the hole left by the pilot probe. The sample probe shall extend at least 3 ft below the landfill cover and shall protrude about 1 ft above the landfill cover. Seal around the sampling probe with bentonite and cap the sampling probe with the sampling probe cap.

4.2.2 Auger Procedure. Use an auger to drill a hole through the landfill cover and to at least 3 ft below the landfill cover. Place the sample probe in the hole and backfill with pea gravel to a level 2 ft from the surface. The sample probe shall protrude at least 1 ft above the landfill cover. Seal the remaining area around the probe with bentonite. Allow 24 hr for the landfill gases to equilibrate inside the augered probe before sampling.

4.3 Sample Train Assembly. Just before assembly, measure the tank vacuum using the pressure gauge. Record the vacuum, the ambient temperature, and the barometric pressure at this time. Assemble the sampling probe purging system as shown in Figure 1.

4.4 Sampling Procedure. Open the sampling valve and use the purge pump and the flow control valve to evacuate at least two sample probe volumes from the system at a flow rate of 100 $\pm$ 10 ml/min. Close the sampling valve and replace the purge pump with the sample tank apparatus as shown in Figure 2. Open the sampling valve and the sample tank valves and, using the flow control valve, sample at a flow rate of 100 $\pm$ 10 ml/min until the sample tank gauge pressure is zero. Disconnect the sampling tank apparatus and use the carrier gas bypass valve to pressurize the sample cylinder to approximately 1,060 mm Hg absolute pressure with helium and record the final pressure.

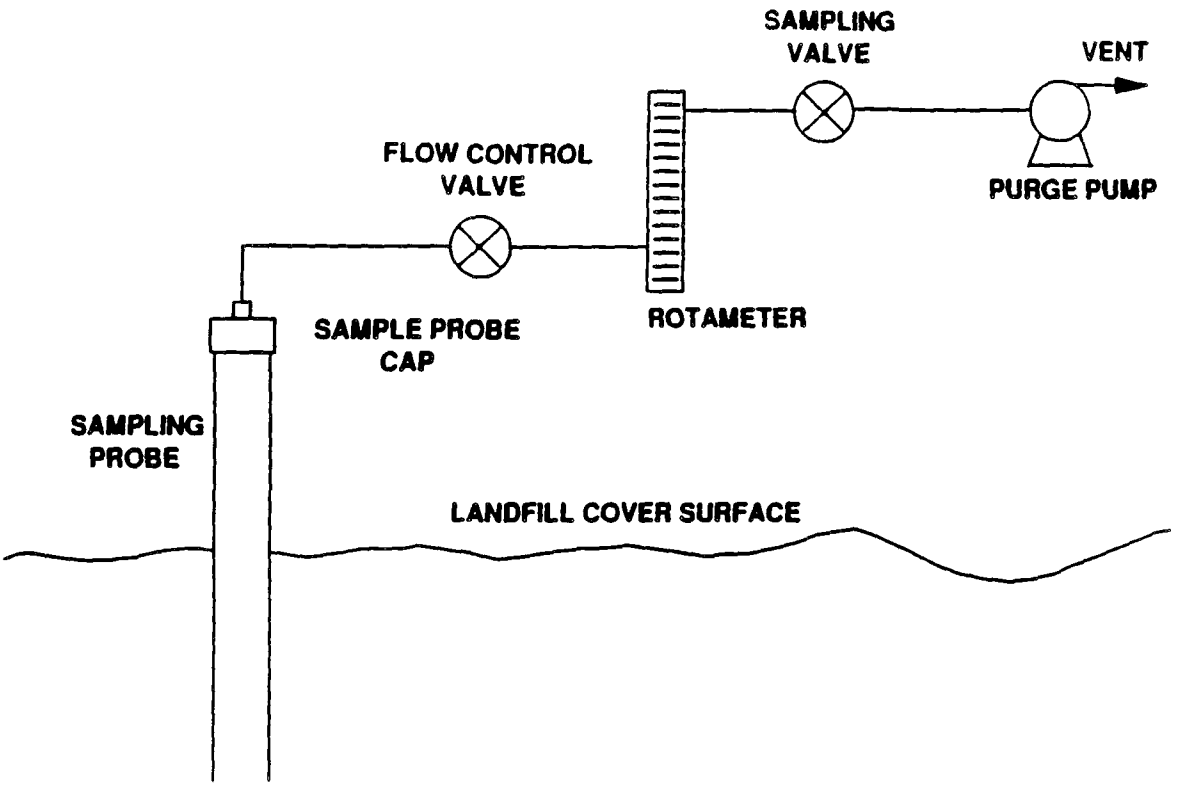

Figure 1. Schematic of sampling probe purging system.

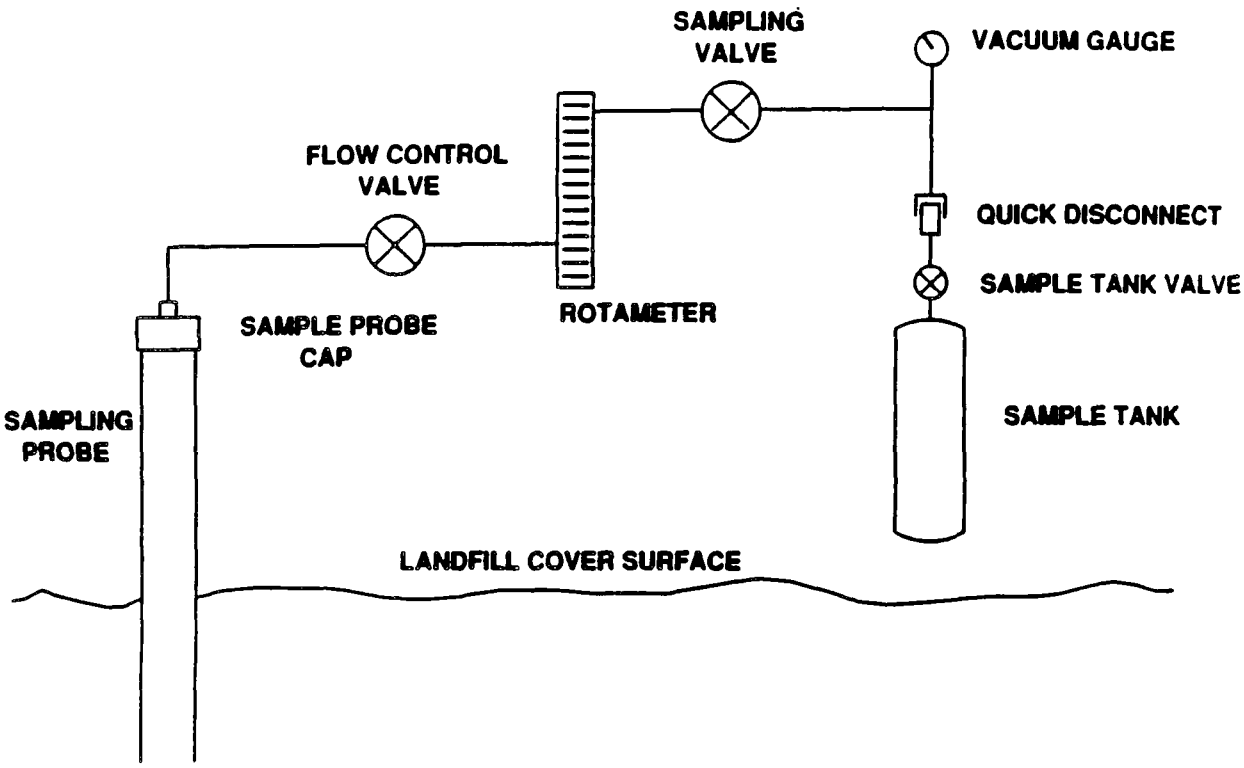

Figure 2. Schematic of sampling train.

Alternatively, the sample tank may be pressurized in the lab. If not analyzing for N_2, the sample cylinder may be pressurized with zero air.

4.4.1 Use Method 3C to determine the percent N_2 in the sample. Presence of N_2 indicates infiltration of ambient air into the gas sample. The landfill sample is acceptable if the concentration of N_2 is less than one percent.

4.5 Analysis. The oxidation, reduction, and measurement of NMOC's is similar to Method 25. Before putting the NMOC analyzer into routine operation, conduct an initial performance test. Start the analyzer, and perform all the necessary functions in order to put the analyzer into proper working order. Conduct the performance test according to the procedures established in Section 5.1. Once the performance test has been successfully completed and the NMOC calibration response factor has been determined, proceed with sample analysis as follows:

4.5.1 Daily Operations and Calibration Checks. Before and immediately after the analysis of each set of samples or on a daily basis (whichever occurs first), conduct a calibration test according to the procedures established in Section 5.2. If the criteria of the daily calibration test cannot be met, repeat the NMOC analyzer performance test (Section 5.1) before proceeding.

4.5.2 Operating Conditions. Same as in Method 25, Section 4.4.2.

4.5.3 Analysis of Sample Tank. Purge the sample loop with sample, and then inject the sample. Under the specified operating conditions, the CO_2 in the sample will elute in approximately 100 seconds. As soon as the detector response returns to baseline following the CO_2 peak, switch the carrier gas flow to backflush, and raise the column oven temperature to 195°C as rapidly as possible. A rate of 30°C/min has

been shown to be adequate. Record the value obtained for any measured NMOC. Return the column oven temperature to 85°C in preparation for the next analysis. Analyze each sample in triplicate, and report the average as C_{tm}.

4.6 Audit Samples. Same as in Method 25, Section 4.5.

5. Calibration and Operational Checks

Maintain a record of performance of each item.

5.1 Initial NMOC Analyzer Performance Test. Same as in Method 25, Section 5.2, except omit the linearity checks for CO_2 standards.

5.2 NMOC Analyzer Daily Calibration.

5.2.1 NMOC Response Factors. Same as in Method 25, Section 5.3.2.

5.3 Sample Tank Volume. The volume of the gas sampling tanks must be determined. Determine the tank volumes by weighing them empty and then filled with deionized water; weigh to the nearest 5 g, and record the results. Alternatively, measure, to the nearest 5 ml, the volume of water used to fill them.

6. Calculations

All equations are written using absolute pressure; absolute pressures are determined by adding the measured barometric pressure to the measured gauge of manometer pressure.

6.1 Nomenclature.

B_w = Moisture content in the sample, fraction.

C_t = Calculated NMOC concentration, ppm C equivalent.

C_{tm} = Measured NMOC concentration, ppm C equivalent.

P_b = Barometric pressure, mm Hg.

P_{ti} = Gas sample tank pressure after evacuation, mm Hg absolute.

P_t = Gas sample tank pressure after sampling, but before pressurizing, mm Hg absolute.

P_{tf} = Final gas sample tank pressure after pressurizing, mm Hg absolute.

P_w = Vapor pressure of H_2O (from Table 1), mm Hg.

T_{ti} = Sample tank temperature before sampling, ^oK.

T_t = Sample tank temperature at completion of sampling, ^oK.

T_{tf} = Sample tank temperature after pressurizing, ^oK.

r = Total number of analyzer injections of sample tank during analysis (where j = injection number, 1...r).

6.2 Water Correction. Use Table 1, the LFG temperature, and barometric pressure at the sampling site to calculate Bw.

$$B_w = \frac{P_w}{P_b}$$

6.3 NMOC Concentration. Use the following equation to calculate the concentration of NMOC for each sample tank.

$$C_t = \left[\frac{\frac{P_{tf}}{T_{tf}}}{\frac{P_t}{T_t} - \frac{P_{ti}}{T_{ti}}} \right] \left[\frac{1}{(1-B_w)\ r} \sum_{j=1}^{r} C_{tm}(j) \right]$$

TABLE 25C-1. MOISTURE CORRECTION

Temperature, ^{o}C	Vapor pressure of H_2O, mm Hg	Temperature, ^{o}C	Vapor pressure of H_2O, mm Hg
4	6.1	18	15.5
6	7.0	20	17.5
8	8.0	22	19.8
10	9.2	24	22.4
12	10.5	26	25.2
14	12.0	28	28.3
16	13.6	30	31.8

7. Bibliography

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APPENDIX F

TABLES ON THE ECONOMIC IMPACTS OF THE ENERGY RECOVERY OPTION

TABLE F-1. SUMMARY INFORMATION FOR AFFECTED CLOSED AND EXISTING LANDFILLS WITH POSITIVE ENERGY RECOVERY COSTS

	Stringency Levels (Mg NMOC/yr)		
	25	100	250
Number of affected landfills	1,024	325	77
(Percent of total closed and existing landfills)	(14)	(5)	(1)
Distribution of affected landfills by design capacity (10^6 Mg)			
≤ 1	470	126	14
	(46)	(39)	(18)
1 to 5	475	170	56
	(46)	(52)	(73)
5 to 10	62	24	2
	(6)	(7)	(3)
> 10	17	5	5
	(2)	(2)	(6)
Total	1,024	325	77
	(100)	(100)	(100)
Privately owned affected landfills	215	68	27
(Percent of affected landfills)	(21)	(21)	(35)
Existing	186	56	17
Closed	29	12	10

Note: The numbers in parentheses are percentages. Details may not add to totals due to rounding.

TABLE F-2. LENGTH OF CONTROL PERIOD FOR AFFECTED CLOSED AND EXISTING LANDFILLS WITH POSITIVE ENERGY RECOVERY COSTS

	Stringency Levels (Mg NMOC/yr)		
	25	100	250
Average length of control period (years)	**69.6**	**50.8**	**36.0**
Distribution of affected landfills by length of control period (years)			
≤ 25	**213** (21)	**167** (51)	**39** (51)
26 to 50	**230** (22)	**49** (15)	**10** (13)
51 to 100	**310** (30)	**36** (11)	**27** (35)
101 to 150	**235** (23)	**51** (16)	**2** (3)
> 150	**36** (4)	**22** (7)	**0** (0.0)
Total	**1024** (100)	**325** (100)	**77** (100)

Note: Numbers in parentheses are percentages. Details may not add to totals due to rounding.

TABLE F-3. LENGTH OF CONTROL PERIOD PRIOR TO CLOSURE FOR AFFECTED EXISTING LANDFILLS WITH POSITIVE ENERGY RECOVERY COSTS

	Stringency Levels (Mg NMOC/yr)		
	25	100	250
Average length of control period prior to closure (years)	**20.9**	**14.5**	**8.6**
Distribution of affected landfills by length of control period prior to closure (years)			
≤ 5	**228** (28)	**124** (49)	**36** (57)
6 to 10	**109** (13)	**36** (14)	**15** (24)
11 to 20	**247** (30)	**53** (21)	**2** (3)
21 to 50	**172** (21)	**10** (4)	**10** (16)
> 50	**65** (8)	**29** (12)	**0** (0)
Total	**821** (100)	**252** (100)	**63** (100)

Note: Numbers in parentheses are percentages. Details may not add to totals due to rounding. Excludes closed landfills.

TABLE F-4. NET PRESENT VALUE OF ENTERPRISE COSTS FOR AFFECTED CLOSED AND EXISTING LANDFILLS: COST-MINIMIZING OPTION AT LANDFILLS WITH POSITIVE ENERGY RECOVERY COSTS

Net Present Value	Stringency Levels (Mg NMOC/yr) 25	100	250
National enterprise costs (10^6)			
Capital	1,052	324	121
Operating	2,024	424	155
Energy Recovery Revenue	1,625	299	154
Total	1,450	450	123
Average total enterprise cost per affected landfill (10^6)	1.42	1.39	1.59
Distribution of affected landfills by net present value of enterprise costs (10^6)			
≤ 0.5	155	61	17
	(15)	(19)	(22)
0.5 to 1.0	179	70	17
	(18)	(22)	(22)
1.0 to 3.0	627	162	33
	(61)	(50)	(43)
3.0 to 5.0	63	32	10
	(6)	(10)	(13)
> 5.0	0	0	0
	(0)	(0)	(0)
Total	1,024	325	77
	(100)	(100)	(100)

Note: Numbers in parentheses are percentages. Net present value of enterprise costs is calculated using a 4 percent discount rate for publicly owned landfills and an 8 percent discount rate for privately owned landfills. Details may not add to totals due to rounding.

TABLE F-5. ANNUALIZED ENTERPRISE CONTROL COST PER Mg OF MSW FOR AFFECTED EXISTING LANDFILLS: COST-MINIMIZING OPTION AT LANDFILLS WITH POSITIVE ENERGY RECOVERY COSTS

	Stringency Level (Mg NMOC/yr)		
	25	100	250
National annualized cost per Mg MSW ($/Mg MSW)	**1.64**	**2.66**	**1.43**
Distribution of affected landfills by annualized cost per Mg MSW ($/Mg MSW)			
≤ 0.50	**104** (13)	**29** (12)	**17** (27)
0.50 to 1.25	**153** (19)	**24** (10)	**0** (0)
1.25 to 3.00	**211** (26)	**90** (36)	**31** (49)
3.00 to 10.00	**259** (32)	**94** (37)	**15** (24)
> 10.00	**94** (11)	**15** (6)	**0** (0)
Total	**821** (100)	**252** (100)	**63** (100)

Note: Numbers in parentheses are percentages. Costs for publicly owned landfills are annualized at 4 percent over the control period. Costs for privately owned landfills are annualized at 8 percent from 1992 to the year of closure. Details may not add to totals due to rounding. Excludes closed landfills.

TABLE F-6. ANNUALIZED ENTERPRISE CONTROL COST PER HOUSEHOLD FOR AFFECTED EXISTING LANDFILLS: COST-MINIMIZING OPTION AT LANDFILLS WITH POSITIVE ENERGY RECOVERY COSTS

	Stringency Level (Mg NMOC/yr)		
	25	100	250
National annualized cost per household ($/Household)	**9.50**	**15.47**	**8.33**
Distribution of affected landfills by annualized cost per household ($/Household)			
≤ 3.50	**138** (17)	**29** (12)	**17** (27)
3.50 to 7.00	**111** (14)	**24** (10)	**0** (0)
7.00 to 15.00	**182** (22)	**83** (33)	**32** (51)
15.00 to 30.00	**162** (20)	**58** (23)	**7** (11)
> 30.00	**228** (28)	**58** (23)	**7** (11)
Total	**821** (100)	**252** (100)	**63** (100)

Note: Numbers in parentheses are percentages. Costs for publicly owned landfills are annualized at 4 percent over the control period. Costs for privately owned landfills are annualized at 8 percent from 1992 to the year of closure. Details may not add to totals due to rounding. Excludes closed landfills.

TABLE F-7. NET PRESENT VALUE OF SOCIAL COSTS FOR AFFECTED CLOSED AND EXISTING LANDFILLS: COST-MINIMIZING OPTION AT LANDFILLS WITH POSITIVE ENERGY RECOVERY COSTS

Net Present Value	Stringency Levels (Mg NMOC/yr) 25	100	250
National social costs ($\$10^6$)			
Capital	2,351	622	239
Operating	2,846	580	213
Energy Recovery Revenue	2,238	374	198
Total	2,958	828	253
Average total social cost per affected landfill ($\$10^6$)	2.89	2.55	3.27
Distribution of affected landfills by net present value of social costs ($\$10^6$)			
≤ 0.5	31	29	7
	(3)	(9)	(9)
0.5 to 1.0	95	29	10
	(9)	(9)	(13)
1.0 to 3.0	530	170	22
	(52)	(52)	(29)
3.0 to 5.0	269	53	14
	(26)	(16)	(18)
5.0 to 10.0	89	44	24
	(9)	(14)	(31)
> 10.0	10	0	0
	(1)	(0)	(0)
Total	1,024	325	77
	(100)	(100)	(100)

Note: Numbers in parentheses are percentages. Net present value of social cost is computed using a two-step discounting procedure. First, capital costs are annualized at 10 percent over the control period. Then, present values are computed by discounting annual operating costs and annualized capital costs at 3 percent. Details may not add to totals due to rounding.

TABLE F-8. SUMMARY INFORMATION FOR AFFECTED NEW LANDFILLS WITH POSITIVE ENERGY RECOVERY COSTS

	Stringency Levels (Mg NMOC/yr)		
	25	100	250
Number of affected landfills	**140**	**39**	**10**
(Percent of total new landfills)	(15)	(4)	(1)
Distribution of affected landfills by design capacity (10^6 Mg)			
≤ 1	**58**	**0**	**0**
	(41)	(0)	(0)
1 to 5	**73**	**32**	**3**
	(52)	(82)	(30)
5 to 10	**7**	**7**	**7**
	(5)	(18)	(70)
> 10	**2**	**0**	**0**
	(1)	(0)	(0)
Total	**140**	**39**	**10**
	(100)	(100)	(100)
Privately owned affected landfills	**34**	**0**	**0**
(Percent of affected landfills)	(24)	(0)	(0)

Note: The numbers in parentheses are percentages. Details may not add to totals due to rounding.

TABLE F-9 LENGTH OF CONTROL PERIOD FOR AFFECTED NEW LANDFILLS WITH POSITIVE ENERGY RECOVERY COSTS

	Stringency Levels (Mg NMOC/yr)		
	25	100	250
Average length of control period (years)	**65.0**	**56.2**	**75.2**
Distribution of affected landfills by length of control period (years)			
≤ 25	**24** (17)	**7** (18)	**3** (30)
26 to 50	**46** (33)	**15** (38)	**0** (0)
51 to 100	**36** (26)	**7** (18)	**7** (70)
101 to 150	**34** (24)	**10** (26)	**0** (0)
Total	**140** (100)	**39** (100)	**10** (100)

Note: Numbers in parentheses are percentages. Details may not add to totals due to rounding.

TABLE F-10. LENGTH OF CONTROL PERIOD PRIOR TO CLOSURE FOR AFFECTED NEW LANDFILLS WITH POSITIVE ENERGY RECOVERY COSTS

	Stringency Levels (Mg NMOC/yr)		
	25	100	250
Average length of control period prior to closure (years)	13.0	12.1	7.3
Distribution of affected landfills by length of control period prior to closure (years)			
≤ 5	29 (21)	7 (18)	3 (30)
6 to 10	24 (17)	0 (0)	7 (70)
11 to 20	73 (52)	32 (82)	0 (0)
21 to 50	14 (10)	0 (0)	0 (0)
Total	140 (100)	39 (100)	10 (100)

Note: Numbers in parentheses are percentages. Details may not add to totals due to rounding.

TABLE F-11. NET PRESENT VALUE OF ENTERPRISE COSTS FOR AFFECTED NEW LANDFILLS: COST-MINIMIZING OPTION AT LANDFILLS WITH POSITIVE ENERGY RECOVERY COSTS

Net Present Value	Stringency Levels (Mg NMOC/yr) 25	100	250
National enterprise costs (10^6)			
Capital	86	64	32
Operating	181	110	48
Energy Recovery Revenue	116	112	62
Total	150	63	18
Average total enterprise cost per affected landfill (10^6)	1.07	1.61	1.83
Distribution of affected landfills by net present value of enterprise costs (10^6)			
≤ 0.5	53	0	0
	(38)	(0)	(0)
0.5 to 1.0	27	7	2
	(19)	(18)	(20)
1.0 to 3.0	53	25	8
	(38)	(64)	(80)
3.0 to 5.0	7	7	0
	(5)	(18)	(0)
>5.0	0	0	0
	(0)	(0)	(0)
Total	140	39	10
	(100)	(100)	(100)

Note: Numbers in parentheses are percentages. Net present value of enterprise costs is calculated using a 4 percent discount rate for publicly owned landfills and an 8 percent discount rate for privately owned landfills. Details may not add to totals due to rounding.

TABLE F-12. ANNUALIZED ENTERPRISE CONTROL COST PER Mg OF MSW FOR AFFECTED NEW LANDFILLS: COST-MINIMIZING OPTION AT LANDFILLS WITH POSITIVE ENERGY RECOVERY COSTS

	Stringency Level (Mg NMOC/yr)		
	25	100	250
National annualized cost per Mg MSW ($/Mg MSW)	**0.95**	**0.92**	**0.59**
Distribution of affected landfills by annualized cost per Mg MSW ($/Mg MSW)			
≤ 0.25	**17** (12)	**0** (0)	**0** (0)
0.25 to 0.50	**10** (7)	**0** (0)	**0** (0)
0.50 to 1.00	**34** (24)	**10** (26)	**8** (80)
1.00 to 3.00	**43** (31)	**29** (74)	**2** (20)
> 3.00	**36** (26)	**0** (0)	**0** (0)
Total	**140** (100)	**39** (100)	**10** (100)

Note: Numbers in parentheses are percentages. Costs for publicly owned landfills are annualized at 4 percent over the control period. Costs for privately owned landfills are annualized at 8 percent over the life of the landfill. Details may not add to totals due to rounding.

TABLE F-13. ANNUALIZED ENTERPRISE CONTROL COST PER HOUSEHOLD FOR AFFECTED NEW LANDFILLS: COST-MINIMIZING OPTION AT LANDFILLS WITH POSITIVE ENERGY RECOVERY COSTS

	Stringency Level (Mg NMOC/yr)		
	25	100	250
National annualized cost per household ($/Household)	5.53	5.36	3.41
Distribution of affected landfills by annualized cost per household ($/Household)			
≤ 0.75	10 (7)	0 (0)	0 (0)
0.75 to 1.50	7 (5)	0 (0)	0 (0)
1.50 to 3.00	10 (7)	0 (0)	8 (80)
3.00 to 10.00	48 (34)	39 (100)	2 (20)
> 10.00	65 (46)	0 (0)	0 (0)
Total	140 (100)	39 (100)	10 (100)

Note: Numbers in parentheses are percentages. Costs for publicly owned landfills are annualized at 4 percent over the control period. Costs for privately owned landfills are annualized at 8 percent over the life of the landfill. Details may not add to totals due to rounding.

TABLE F-14. NET PRESENT VALUE OF SOCIAL COSTS FOR AFFECTED NEW LANDFILLS: COST-MINIMIZING OPTION AT LANDFILLS WITH POSITIVE ENERGY RECOVERY COSTS

Net Present Value	Stringency Levels (Mg NMOC/yr) 25	100	250
National social costs (10^6)			
Capital	261	146	77
Operating	326	151	69
Energy Recovery Revenue	278	155	88
Total	309	142	58
Average total social cost per affected landfill (10^6)	2.20	3.68	5.95
Distribution of affected landfills by net present value of social costs (10^6)			
≤ 0.5	0 (0)	0 (0)	0 (0)
0.5 to 1.0	24 (17)	0 (0)	0 (0)
1.0 to 3.0	82 (59)	22 (56)	2 (20)
3.0 to 5.0	27 (19)	10 (26)	0 (0)
> 5.0	7 (5)	7 (18)	8 (80)
Total	140 (100)	39 (100)	10 (100)

Note: Numbers in parentheses are percentages. Net present value of social cost is computed using a two-step discounting procedure. First, capital costs are annualized at 10 percent over the control period. Then, present values are computed by discounting annual operating costs and annualized capital costs at 3 percent. Details may not add to totals due to rounding

TABLE F-15. NET PRESENT VALUE OF EMISSIONS REDUCTIONS FOR AFFECTED CLOSED AND EXISTING LANDFILLS: COST-MINIMIZING OPTION AT LANDFILLS WITH POSITIVE ENERGY RECOVERY COSTS

	Stringency Levels (Mg NMOC/yr)		
Net Present Value	**25**	**100**	**250**
Undiscounted NMOC emission reduction (10^6 Mg)	**5.81**	**3.06**	**1.26**
Discounted NMOC emission reduction (10^6 Mg)	**2.04**	**1.15**	**0.59**
Average discounted NMOC emission reduction per affected landfill (Mg)	**1,993**	**3,546**	**7,560**
Distribution of affected landfills by discounted NMOC emission reduction per affected landfill (Mg)			
≤ 1,000	**429** (42)	**82** (25)	**17** (22)
1,000 to 2,000	**305** (30)	**94** (29)	**0** (0)
2,000 to 5,000	**208** (20)	**94** (29)	**22** (29)
5,000 to 10,000	**58** (6)	**29** (9)	**14** (18)
> 10,000	**24** (2)	**24** (8)	**24** (31)
Total	**1,024** (100)	**324** (100)	**77** (100)

Note: Numbers in parentheses are percentages. Net present value of emission reductions is calculated using a 3 percent discount rate. Details may not add to totals due to rounding.

TABLE F-16. COST EFFECTIVENESS FOR AFFECTED CLOSED AND EXISTING LANDFILLS: COST-MINIMIZING OPTION AT LANDFILLS WITH POSITIVE ENERGY RECOVERY COSTS

	Stringency Level (Mg NMOC/yr)		
	25	100	250
National cost effectiveness ($/Mg NMOC)	1,449	719	433
Distribution of affected landfills by cost effectiveness ($/Mg NMOC)			
≤ 1,000	157 (15)	189 (58)	68 (88)
1,000 to 2,000	269 (26)	102 (31)	7 (9)
2,000 to 5,000	414 (41)	7 (2)	0 (0)
5,000 to 10,000	143 (14)	12 (4)	2 (3)
> 10,000	41 (4)	15 (5)	0 (0)
Total	1,024 (100)	325 (100)	77 (100)
Incremental cost effectiveness	2,287	989	—

Note: Numbers in parentheses are percentages. Cost effectiveness is calculated by dividing the net present value of social cost by the discounted NMOC emission reduction (see Tables F-7 and F-15). Details may not add to totals due to rounding.

TABLE F-17. NET PRESENT VALUE OF EMISSIONS REDUCTIONS FOR AFFECTED NEW LANDFILLS: COST-MINIMIZING OPTION AT LANDFILLS WITH POSITIVE ENERGY RECOVERY COSTS

	Stringency Levels (Mg NMOC/yr)		
Net Present Value	**25**	**100**	**250**
Undiscounted NMOC emission reduction (10^6 Mg)	**0.83**	**0.49**	**0.25**
Discounted NMOC emission reduction (10^6 Mg)	**0.25**	**0.15**	**0.06**
Average discounted NMOC emission reduction per affected landfill (Mg)	**1,765**	**3,818**	**6,680**
Distribution of affected landfills by discounted NMOC emission reduction per affected landfill (Mg)			
≤ 1,000	**77** (55)	**0** (0)	**0** (0)
1,000 to 2,000	**17** (12)	**7** (18)	**2** (20)
2,000 to 5,000	**39** (28)	**25** (64)	**0** (0)
> 5,000	**7** (5)	**7** (18)	**8** (80)
Total	**140** (100)	**39** (100)	**10** (100)

Note: Numbers in parentheses are percentages. Net present value of emission reductions is calculated using a 3 percent discount rate. Details may not add to totals due to rounding.

TABLE F-18. COST EFFECTIVENESS FOR AFFECTED NEW LANDFILLS: COST-MINIMIZING OPTION AT LANDFILLS WITH POSITIVE ENERGY RECOVERY COSTS

	Stringency Level (Mg NMOC/yr)		
	25	100	250
National cost effectiveness ($/Mg NMOC)	**1,244**	**963**	**891**
Distribution of affected landfills by cost effectiveness ($/Mg NMOC)			
≤ 1,000	**24** (17)	**15** (38)	**7** (70)
1,000 to 2,000	**53** (38)	**24** (64)	**3** (30)
2,000 to 5,000	**39** (28)	**0** (0)	**0** (0)
5,000 to 10,000	**17** (12)	**0** (0)	**0** (0)
> 10,000	**7** (5)	**0** (0)	**0** (0)
Total	**140** (100)	**39** (100)	**10** (100)
Incremental cost effectiveness	**1,661**	**870**	**—**

Note: Numbers in parentheses are percentages. Cost effectiveness is calculated by dividing the net present value of social cost by the discounted NMOC emission reduction (see Tables F-14 and F-17). Details may not add to totals due to rounding.

APPENDIX G

THEORETICAL COLLECTION SYSTEM DESIGN

APPENDIX G

THEORETICAL COLLECTION SYSTEM DESIGN

G.1 INTRODUCTION

This appendix provides the theoretical approach for designing landfill gas collection systems. Design equations for active vertical wells, active horizontal trenches, and passive vertical wells are detailed in Sections G.3, G.4, and G.5, respectively. These equations were used in Chapters 5, 6, and 7 to quantify the nationwide impact of controlling landfills and as the foundation for the collection system design procedure outlined in Chapter 9. The design procedure in Chapter 9 is a graphical interpretation of the theoretical design equation. The derivation of this procedure, is provided in Section G.6.

G.2 ASSUMPTIONS

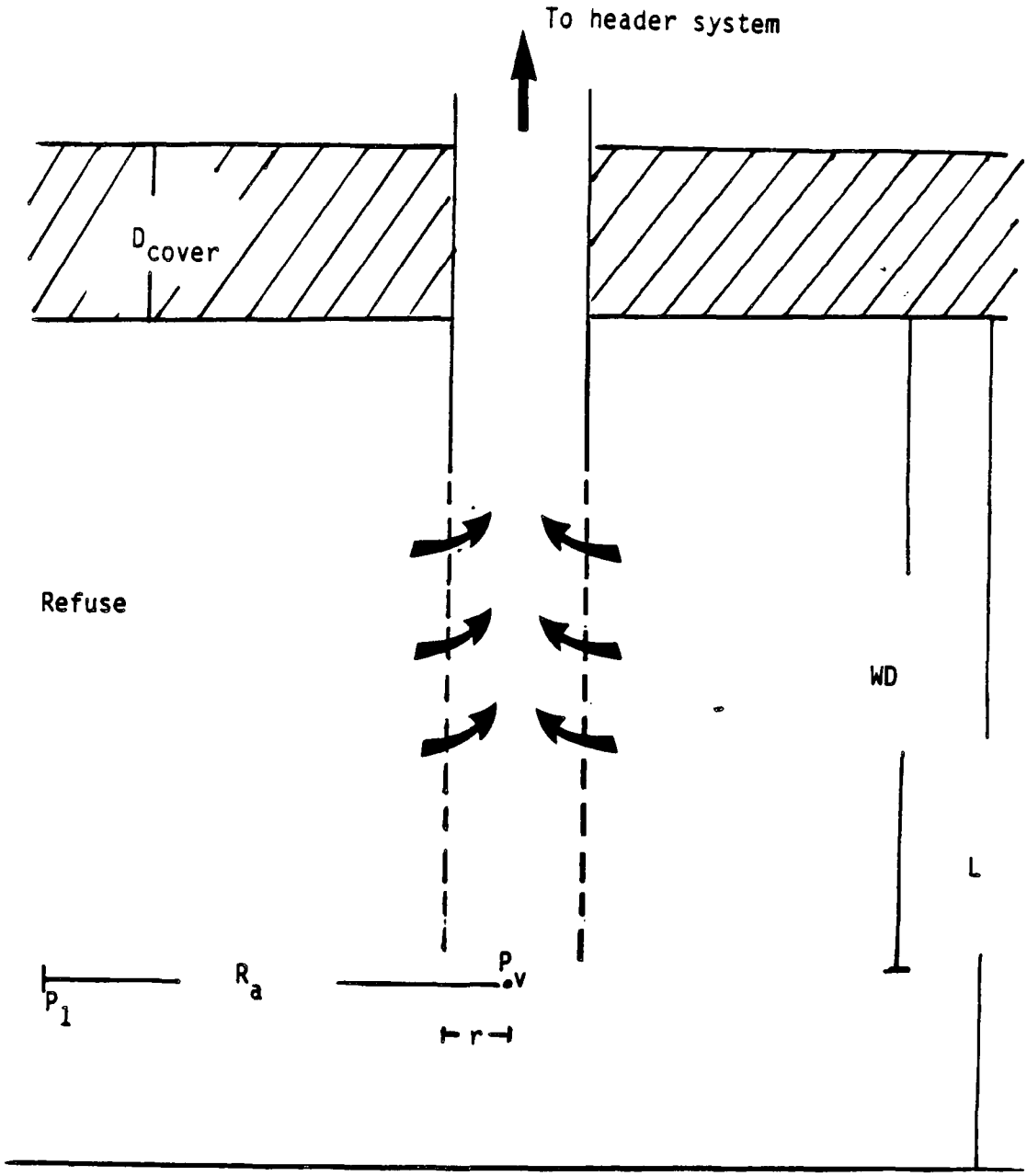
The following assumptions have been made in developing the design equations for landfill gas collection systems:

- The design of the active vertical and passive collection systems is based on the peak landfill gas generation rate which is calculated using: (1) an equation that describes the radius of influence of extraction wells and (2) site-specific information for each landfill (e.g., amount of refuse in place, landfill depth, landfill age, acceptance rate, etc.).
- Scholl Canyon Model, a first order decay model described in Chapter 3, is used to estimate the landfill gas generation rate.
- The lag time (typically less than one to two years) for the landfill gas generation is negligible when compared to the total life of landfill gas generation. Thus, the peak landfill gas generation rate is assumed to occur at the time of closure.

G.3 THEORETICAL APPROACH FOR ACTIVE VERTICAL WELL COLLECTION SYSTEM DESIGN

The geometry of an active well system is illustrated in Figure G-1. The radius of influence for a vertical well can be obtained by the following mass balance equation:

$$R_a = (Q_{w,a} \text{ Design Capacity}/\pi L \, \rho_{refuse} \, Q_{gen} \, E_a)^{1/2} \qquad (1)$$



SIDE VIEW

R_a = radius of influence
D_{cover} = cover thickness
WD = well depth
L = landfill depth
P_v = vacuum pressure
P_1 = internal landfill pressure
r = radius of well

Figure G-1. Model active vertical well collection system geometry.

where,

R_a = radius of influence for active collection systems, m

$Q_{w,a}$ = landfill gas flowrate per well, m^3/sec

Design Capacity = design capacity of the landfill, kg

π = 3.14

refuse = refuse density, kg/m^3

L = landfill depth, m

Q_{gen} = peak landfill gas generation rate, m^3/sec

E_a = fractional collection efficiency of active well systems

Equation (1) calculates the radius of influence based on the maximum landfill gas generation rate (Q_{gen}) and the collection efficiency of the active vertical well system (E_a). If the lag time for landfill gas generation is neglected, Q_{gen} is assumed to occur at the time of landfill closure and can be determined using the Scholl Canyon model:

$$Q_{gen} = 2 \ L_o \ R \ (1 - \exp(-kt)) \qquad (2)$$

where,

Q_{gen} = peak landfill gas generation rate, m^3/yr

L_o = refuse methane generation potential, m^3 methane/Mg refuse

R = average refuse acceptance rate, Mg/yr

k = landfill gas generation rate constant, 1/yr

t = landfill age upon closure

To calculate Q_{gen} using Equation (2), it is necessary to know values for L_o and k. As discussed above, L_o and k vary from landfill to landfill depending on the composition, moisture content, pH, and internal landfill temperature. Values of L_o and k have been determined empirically for a total of 54 landfills based on test well data and/or data from existing landfill gas collecting systems.[1] For these landfills, the estimated L_o and k correspond to the collected landfill gas flowrate

(Q_{gen} x E_a) rather than the total landfill gas generation rate. Using the values of L_o and k derived in this way, the product of Q_{gen} and E_a may be calculated using the following equation:

$$Q_{gen} E_a = 2 L_o' R [1 - \exp(-k't)] \quad (3)$$

where,

L_o' = refuse methane generation potential estimated from test well data and/or existing landfill gas collection system, m^3 methane/Mg refuse.

k' = landfill gas generation rate constant estimated from test well data and/or existing landfill gas collection system, 1/yr

Once the radius of influence is calculated, the number of wells necessary can be calculated from the landfill area.

$$n = A/(\pi R_a^2) \quad (4)$$

where,

n = number of wells
A = area of landfill, m^2
= design capacity/(refuse density X depth)
R_a = radius of influence, m
π = 3.14

From Darcy's Law, the landfill pressure corresponding to the calculated radius of influence, refuse permeability, the magnitude of vacuum applied, and the collectable landfill gas flowrate (i.e. Q_{gen} X E_a) can be calculated.[2]

$$\frac{P_1^2 - P_v^2}{P_v} = \frac{R_a^2 \ln(R_a/r) \mu_{lfg} \quad refuse (Q_{gen} E_a)}{\text{Design Capacity } k_{refuse} (WD/L)} \quad (5)$$

where,

P_1 = internal landfill pressure, Newton/m^2
P_v = vacuum pressure, Newton/m^2

R_a = radius of influence, m

r = radius of outer well (or gravel casing), m

refuse = refuse density, 650 kg/m^3

k_{refuse} = intrinsic refuse permeability, m^2

μ_{lfg} = landfill gas viscosity, Newton-sec/m^2

Design Capacity = design capacity of landfill, kg

WD = well depth (i.e., 0.75L), m

L = landfill depth, m

Q_{gen} = peak landfill gas generation rate, ft^3/yr

E_a = fractional collection efficiency of active well system

Once the radius of influence and the number of wells are calculated, it is necessary to check if significant air infiltration exists under the given refuse permeability, cover permeability, and vacuum applied.

The flow of air through the cover material is illustrated in Figure G-2. At steady state, the flowrate through the interface of atmosphere and the cover material, and the flowrate through the interface of cover material and the refuse are the same. Thus, the following equation is obtained at steady state:

$$v_{air} = k_{cover} (P_{atm} - P_i)/(\mu_{air} D_{cover}) \quad (6)$$
$$= k_{refuse,v} (P_i - P_v)/(\mu_{air} X)$$

where,

v_{air} = air velocity through cover and refuse, m/sec

k_{cover} = intrinsic cover permeability, m^2

P_{atm} = atmospheric pressure, Newton/m^2

P_i = interface pressure, Newton/m^2

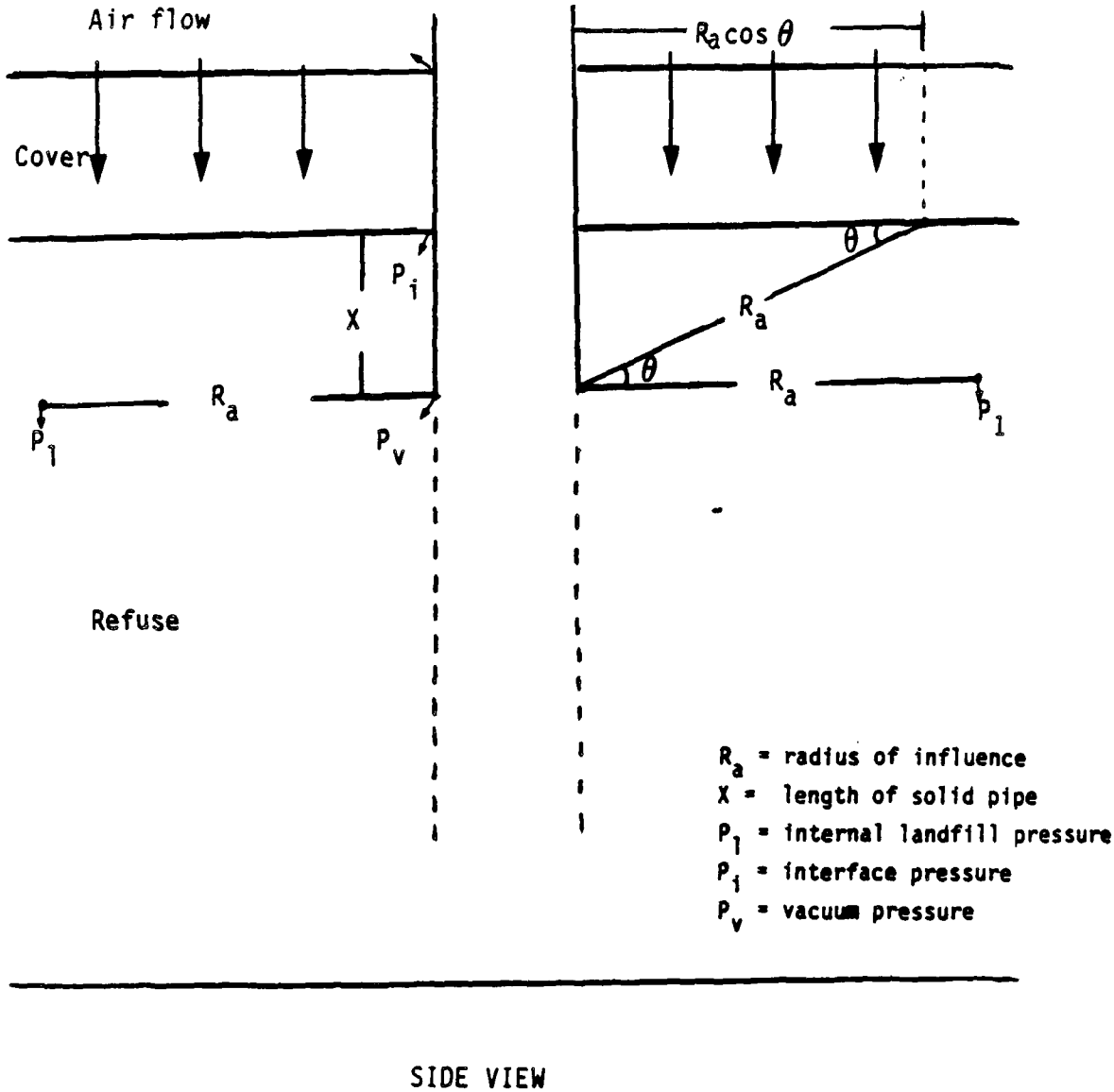
μ_{air} = air viscosity, Newton-sec/m^2

D_{cover} = cover thickness, m

$k_{refuse,v}$ = intrinsic vertical refuse permeability, Newton-sec/m^2

P_v = vacuum pressure, Newton/m^2

X = length of solid pipe, m



SIDE VIEW

Figure G-2. Air flow through landfill cover.

It should be noted that the vertical refuse permeability is used for air infiltration equations rather than the horizontal permeability (or simply permeability). According to industry experts, the horizontal permeability is approximately 10 times greater than the vertical permeability due to the layering effect of the refuse accumulation.[3]

The flowrate of air can be calculated using the following equation:

$$Q_{air} = (v_{air})\ (R_a \cos(\theta))^2$$
$$= v_{air}\ R_a^2\ (\text{if } \theta = \sim 0) \qquad (7)$$

If the maximum allowable percent of oxygen in the total collected landfill gas is assumed to be 0.5 percent, the corresponding allowable percent of air in landfill gas is 2.44 percent.[4] Therefore, the minimum solid pipe length required (X) can be calculated by the following equation:

$$(0.0244)(Q_{gen}\ E_a) = k_{cover}\ (P_{atm} - P_i)\ A/(\mu_{air}\ D_{cover}) \qquad (8)$$
$$= k_{refuse,v}\ (P_i - P_v)\ A\ /(\mu_{air}\ X)$$

Note that Equation (8) only accounts for the air infiltration from the surface of a landfill (i.e, the air infiltration from the sides of landfill is negligible compared to the air infiltration from the surface of landfill). Equation (8) can be simplified to:

$$X = \{[k_{refuse,v}\ k_{cover}\ (P_{atm} - P_v)\ A/\mu_{air}\ (0.0244)(Q_{gen}\ E_a)] - k_{refuse,v}\ D_{cover}\}/k_{cover} \qquad (9)$$

If the required solid pipe length is greater than the available solid pipe length (based on the given landfill depth), the landfill is considered shallow and the magnitude of vacuum needs to be reduced to meet the 2.44 percent air content requirement. The available solid pipe length can be estimated by assuming that the well depth is 75 percent of the landfill depth and two thirds of the well depth needs to be perforated and one third of the

well needs to be solid.[5] For shallow landfills, the magnitude of vacuum required can be calculated using Equation (9) by setting X to be the available solid pipe length.

The radius of influence is then recalculated based on the new vacuum and the landfill pressure calculated using Equation (5). The radius of influence for shallow landfills is expected to be smaller since the pressure driving force (or pressure gradient) would be less. Thus, to achieve the same collection efficiency in a shallow landfill as in a deeper landfill, the number of wells required in a shallow landfill will be larger.

The design calculation steps for active vertical well collections systems are illustrated in Figure G-3.

G.4 THEORETICAL APPROACH FOR HORIZONTAL TRENCH COLLECTION SYSTEMS DESIGN

The geometry of a model horizontal trench system is illustrated in Figure G-4. The governing equations for horizontal trench systems are also based on a mass balance equation and Darcy's Law. The basic approach for designing horizontal trench collection systems is to use the radius of influence calculated for active vertical wells (using Equation (1)) to determine the horizontal spacing between trenches, since the radius of influence is a function of the refuse permeability and the landfill pressure. The landfill pressure, in turn, is a function of the landfill gas generation rate and degree of containment (i.e., type of liner, etc.). The vertical spacing between the trench layers can be calculated by the following equations using vertical refuse permeability.

$$R_v^2 \ln(R_v/r) = [(P_1^2 - P_v^2) \text{ Design Capacity } k_{refuse,v} (WD/L)]/ [P_v \mu_{LFG} \text{ refuse } (Q_{gen} E_a)] \quad (11)$$

$$S_v = 2 R_v$$

where,

P_1 = internal landfill pressure, Newton/m^2

P_v = average vacuum pressure along the trench length, Newton/m^2

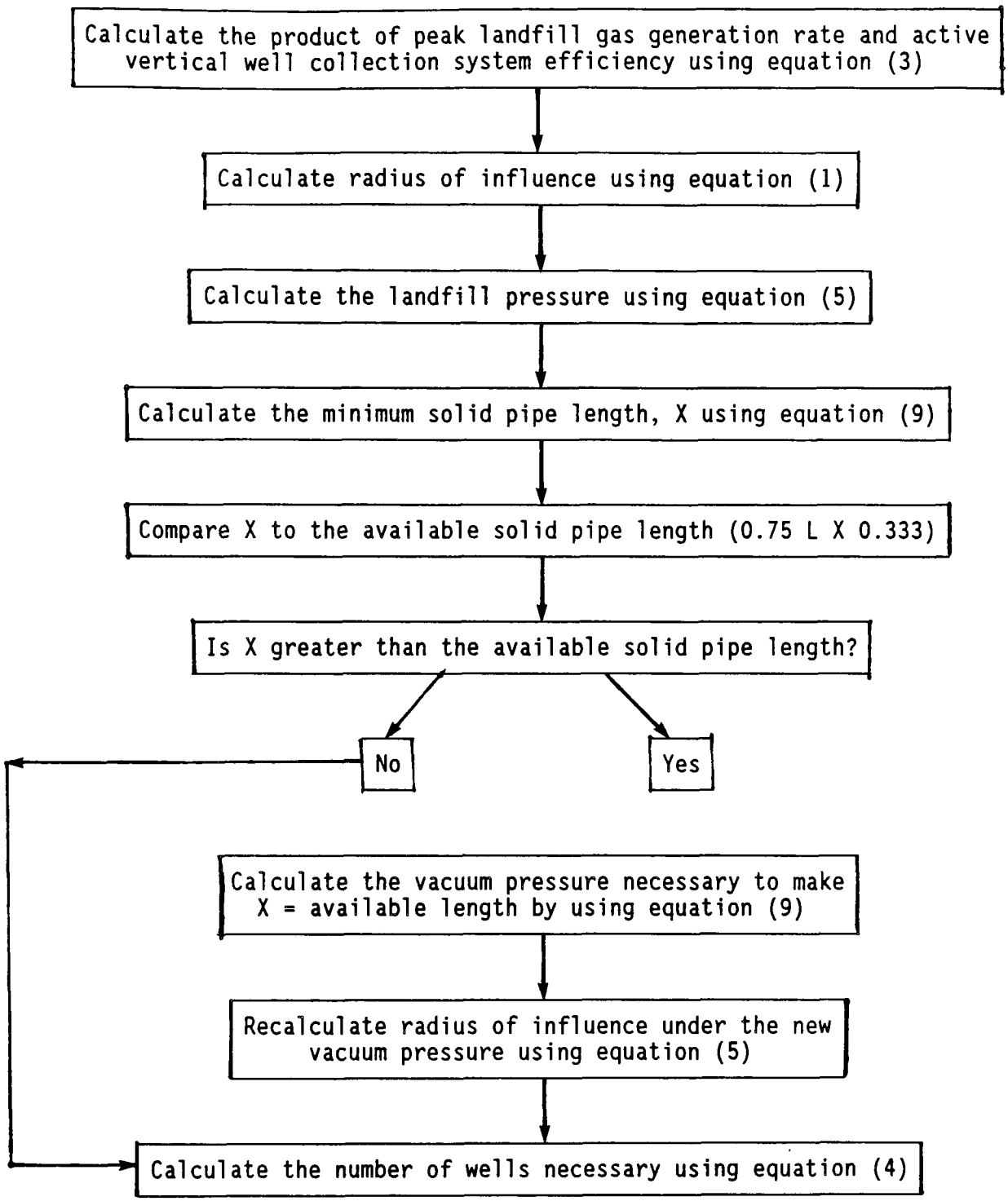


Figure G-3. Active vertical well collection system design calculation steps.

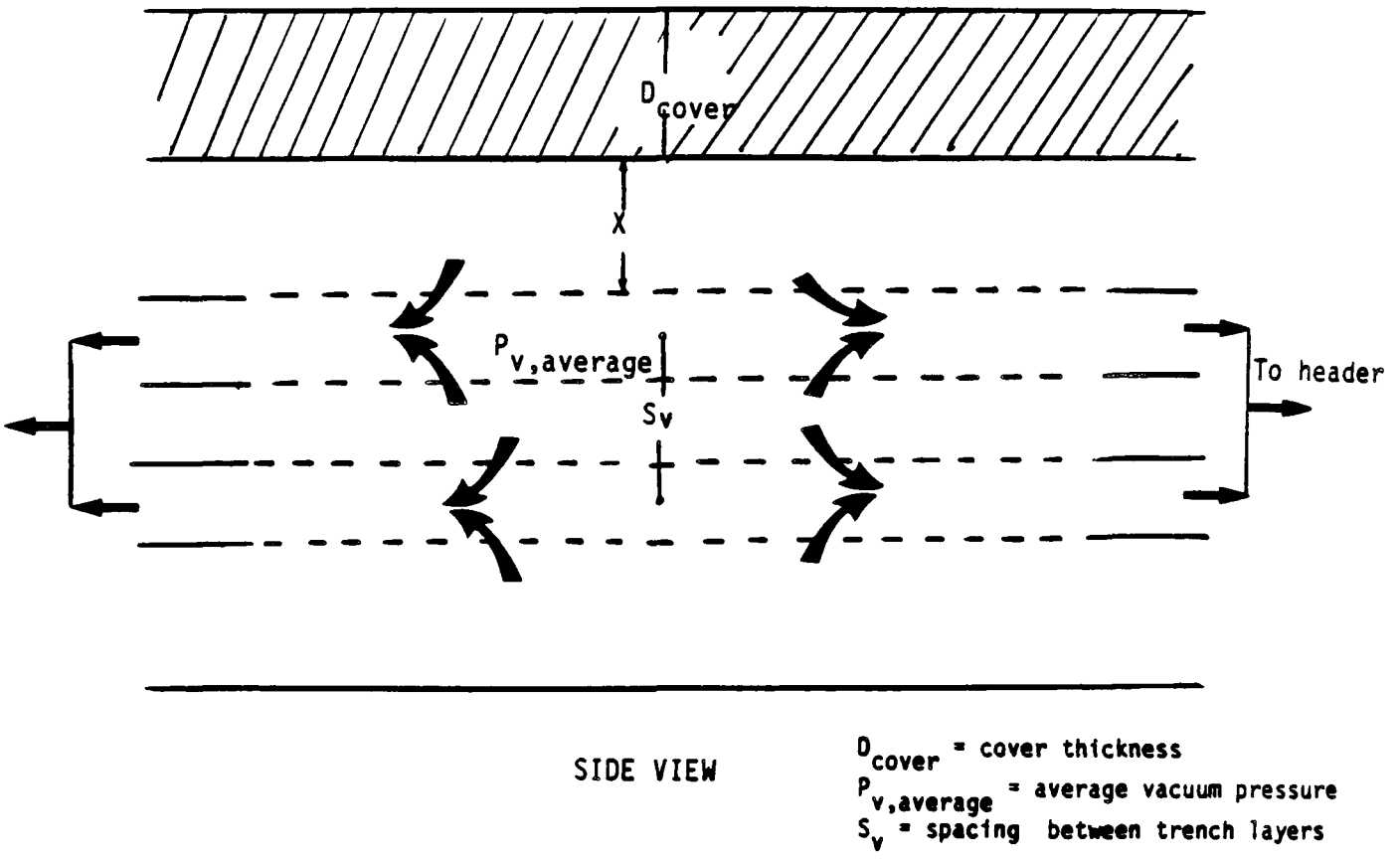


Figure G-4. Model horizontal trench system geometry.

S_v = vertical spacing between trench layers (i.e., radius of influence for vertical direction), m

R_v = vertical radius of influence, m

r = radius of gravel casing, m

$_{refuse}$ = refuse density, 650 kg/m^3

μ_{lfg} = landfill gas viscosity, Newton-sec/m^2

$k_{refuse,v}$ = intrinsic vertical refuse permeability, m^2

Design Capacity = design capacity of landfill, kg

WD = well depth, m (typically 0.75 L)

L = landfill depth, m

Q_{gen} = peak landfill gas generation rate, m^3/yr

E_a = fractional collection efficiency of active well system

Note that the vacuum pressure used in Equation (11) is an average vacuum pressure along the length of a trench. If the vacuum is pulled only at one end of a trench, there may be a significant pressure drop along the length of the trench unless the collected gas flowrate is too small to yield a significant pressure drop. The pressure drop can be minimized if vacuum is pulled evenly using a manifold system.

The number of trench layers can be calculated by:

$$n_l = L/S_v \tag{12}$$

where,

n_l = number of trench layers

L = landfill depth, m

S_v = vertical spacing between trenches, m

Once the vertical spacing between the trench layers is calculated, the horizontal spacing between trenches can be calculated by the following equations:

$$R_h^2 \ln(R_h/r) = [(P_1^2 - P_v^2) \text{ Design Capacity } k_{refuse,h} (WD/L)]/ [P_v \mu_{lfg} \rho_{refuse} (Q_{gen} E_a)] \quad (13)$$

$$S_h = 2 R_h$$

where,

P_1 = internal landfill pressure, Newton/m^2

P_v = average vacuum pressure along the trench length, Newton/m^2

R_h = horizontal radius of influence, m

S_h = horizontal spacing between trench layers, m

r = radius of gravel casing, m

μ_{lfg} = landfill gas viscosity, Newton-sec/m^2

ρ_{refuse} = refuse density, 650 kg/m^3

$k_{refuse,h}$ = intrinsic horizontal refuse permeability, m^2

Design Capacity = design capacity of landfill, kg

WD = well depth, m (typically 0.75 L)

L = landfill depth, m

Q_{gen} = peak landfill gas generation rate, m^3/yr

E_a = fractional collection efficiency of active well system

Assuming that the landfill is square, the number of trenches per trench layer can be calculated by:

$$n_t = A^{1/2} / S_h \quad (14)$$

where,

n_t = number of trenches per trench layer

A = landfill area, m^2

S_h = horizontal spacing between trenches, m

Therefore, the total required trench length for a square landfill is:

$$L_t = n_t \, n_l \, A^{1/2} \tag{15}$$

where,

L_t = total length of trench, m
A = landfill area, m^2

The air infiltration equations for the active vertical collection systems also apply to the horizontal trench collection systems. If the landfill is shallow, the radii of influence for vertical and horizontal directions are calculated (for active vertical well systems) using the reduced magnitude of vacuum and they are applied to horizontal trench systems as the vertical and horizontal spacings.

The design calculation steps for horizontal trench collection systems are presented in Figure G-5.

G.5 THEORETICAL APPROACH FOR PASSIVE COLLECTION SYSTEMS DESIGN

The geometry of the model passive well system is illustrated in Figure G-6. The governing equations for active systems also apply to passive systems except that the pressure gradient in Equation (5) is based on the difference in landfill pressure and atmospheric pressure as follows:

$$\frac{P_l^{\,2} - P_{atm}^{\,2}}{P_{atm}} = \frac{R_p^{\,2} \ln(R_p/r) \, \mu_{lfg} \, \rho_{refuse} \, (Q_{gen} \, E_p)}{\text{Design Capacity } k_{refuse} \, (WD/L)} \tag{16}$$

where,

P_l = internal landfill pressure, Newton/m^2
P_{atm} = atmospheric pressure, Newton/m^2
R_p = radius of influence for passive system, m
r = radius of outer well (or gravel casing), m
refuse = refuse density, 650 kg/m^3
k_{refuse} = intrinsic refuse permeability, m^2
μ_{lfg} = landfill gas viscosity, Newton-sec/m^2

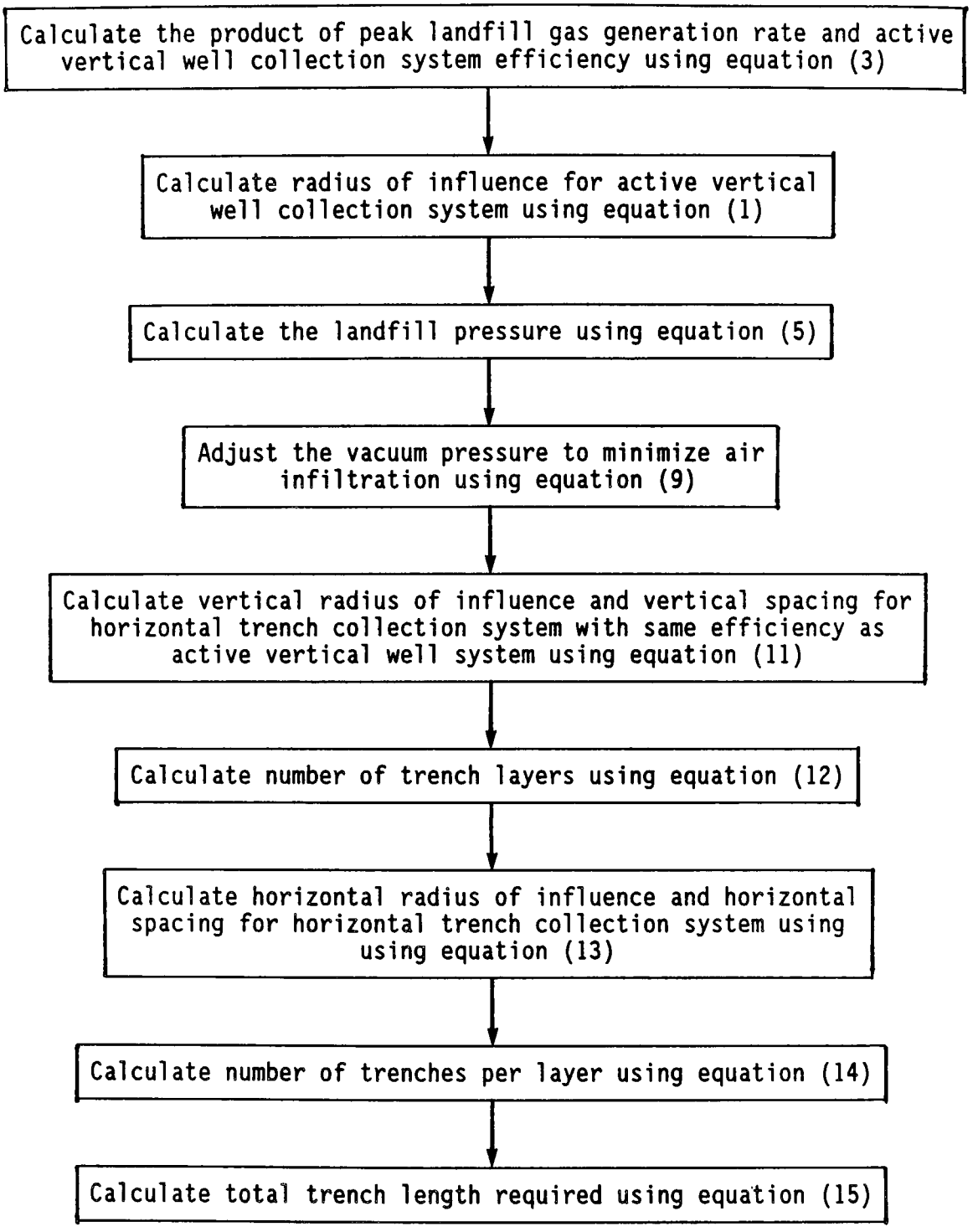


Figure G-5. Horizontal trench system design calculation steps.

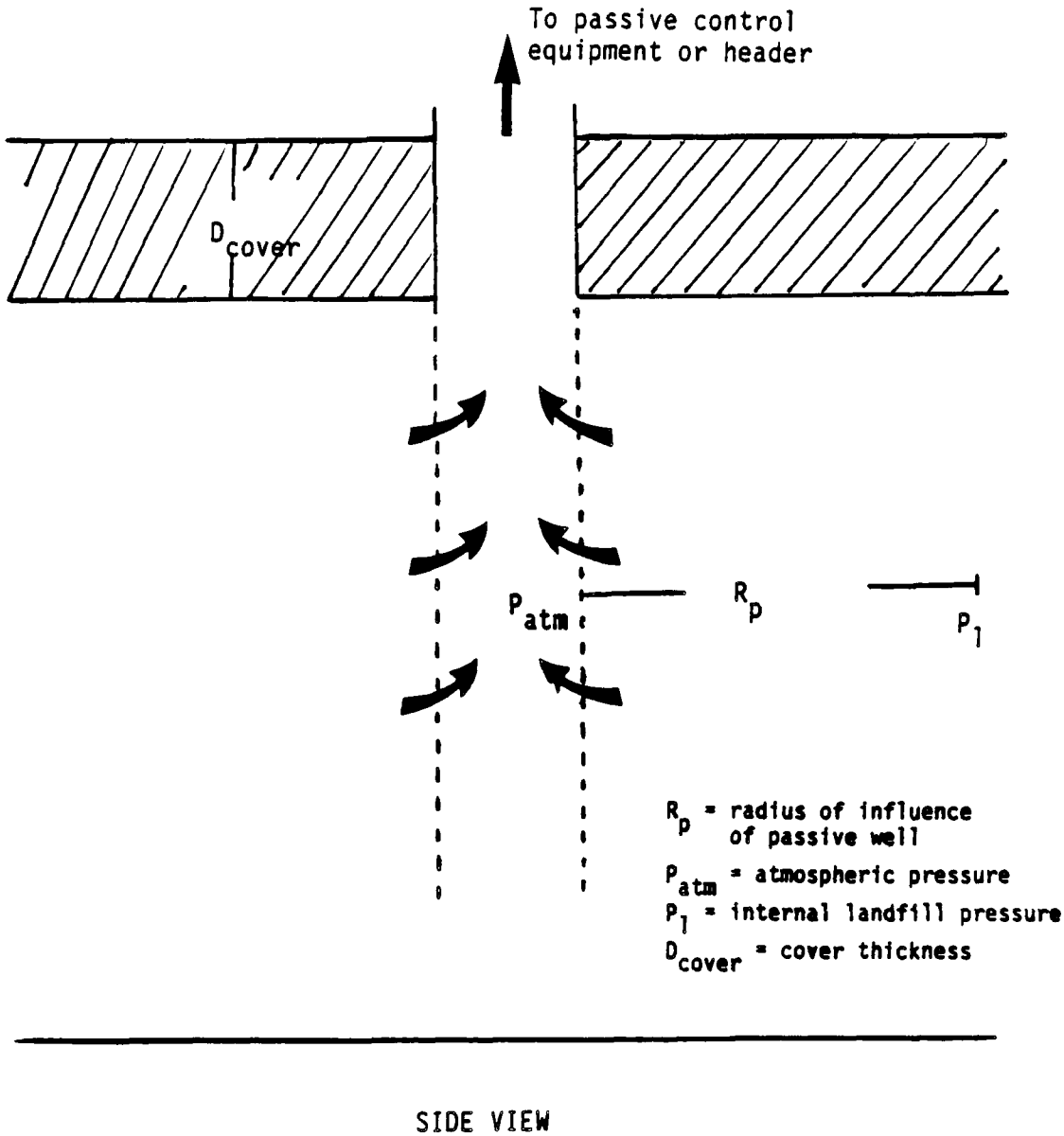


Figure G-6. Model passive collection system geometry.

Q_{gen} = peak landfill gas generation rate, m^3/yr

E_a = fractional collection efficiency of passive well system

Design Capacity = design capacity of landfill, kg

WD = well depth, m (typically 0.75 L)

L = landfill depth, m

The ratio of the radius of influence of passive systems to the radius of influence of active systems can be expressed by the following equation:

$$\frac{R_p^2 \ln(R_p/r)}{R_a^2 \ln(R_a/r)} = \frac{[(P_1^2 - P_{atm}^2)/P_{atm}]\ E_a}{[(P_1^2 - P_v^2)/P_v]\ E_p} \tag{17}$$

By setting the ratio of collection efficiencies on passive systems and active systems to one, the passive system design needed to achieve the same collection efficiency as an active system can be determined. Based on the radius of influence of the passive wells obtained from Equation (17) the number of passive wells necessary can be calculated as follows:

$$n = A/(\pi R_p^2) \tag{18}$$

where,

n = number of wells

A = landfill area, m^2

R_p = radius of influence for passive system, m

As discussed earlier, the problem of air infiltration does not exist for passive systems since the passive systems rely on the natural pressure gradient. The design calculation steps for passive collection systems are illustrated in Figure G-7.

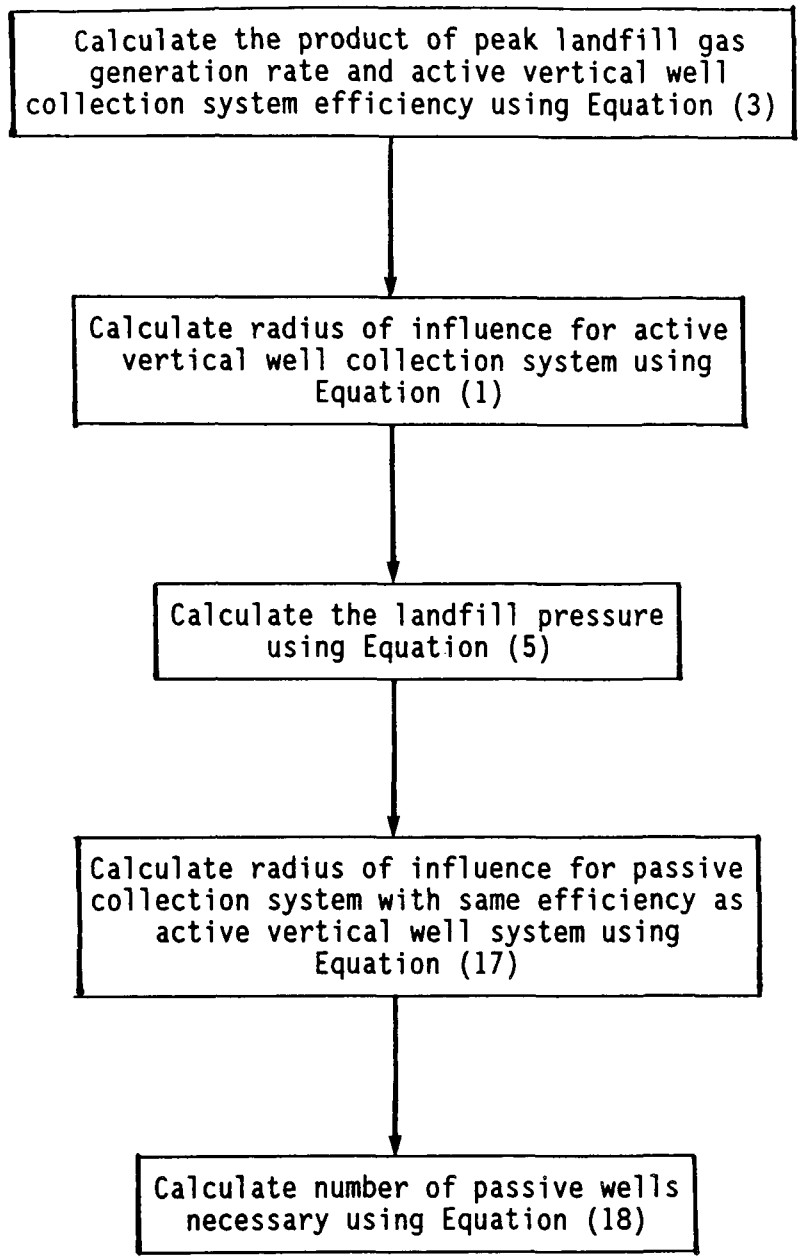


Figure G-7. Passive collection system design calculation steps

G.6 GRAPHICAL INTERPRETATION OF THE THEORETICAL APPROACH

A graphical interpretation of the design equations provided in the previous sections was performed to simplify the approach landfill owners would have to take to design collection systems in the absence of site-specific data. Sections G.6.1 and G.6.2 describe the derivation of the simplified design approach for active collection systems and passive collection systems, respectively.

G.6.1 Simplified Approach for Active Collection System Design

The approach outlined in Chapter 9 for active collection systems is a two step process. The first step is to determine the maximum blower vacuum allowed for a given landfill depth. From Equation 9 in Section G.3, a relationship between the blower vacuum (P_v) and the landfill depth (L) was obtained.

- Derivation of P_v as a function of L

  Given: Equation 9

$$x = \frac{\dfrac{(k_{refuse,v})(k_{cover})(P_{atm} - P_v)\ A}{\mu_{air}\ .0244\ Q_{gen}} - (k_{refuse,v})(D_{cover})}{k_{cover}}$$

From the well specifications, x, the length of solid pipe, is equal to 1/3 the well depth which is 75 percent of the landfill depth.

$$\therefore\ .25L = \frac{\dfrac{(k_{refuse,\ v})(k_{cover})(P_{atm} - P_v)\ A}{\mu_{air}\ .0244\ Q_{gen}} - (k_{refuse,v})(D_{cover})}{k_{cover}}$$

Solving for P_v:

$$P_v = P_{atm} - [(.25L)(k_{cover}) + (k_{refuse})(D_{cover})] * (Q_{gen}/A)(.0244/k_{cover})(\mu_{air}/k_{refuse})$$

But A can be expressed in terms of L

$$A = \frac{DC}{\rho_{refuse} L}$$

where,

L = Landfill depth, m

DC = Design Capacity, Mg

$_{refuse}$ = refuse density, kg/m^3

$$\therefore P_v = P_{atm} - [(.25L)(k_{cover}) + (k_{refuse})(D_{cover})] * (Q_{gen/DC})(\ _{refuse}L)(.0244/k_{cover})(\ _{air}/k_{refuse})$$

Using the following values for refuse density, refuse permeability, and air viscosity:

$_{refuse}$ = 650 kg/m^3

k_{refuse} = 3.743 x 10^{-13} m^2

$_{air}$ = 1. 8 x 10^{-5} N-sec/m^2

and assuming atmospheric pressure is equal to 1 atm, the equation becomes:

$$P_v = 1 - [(.25L)(k_{cover}) + (D_{cover})(3.743 \times 10^{-13})] * (Q_{gen}/DC)(L/k_{cover})(.004)$$

The ratio of Q_{gen} to DC will vary from landfill to landfill due to differences in active life and refuse composition. For the sake of simplicity, however, a single conservative value of this ratio was developed and used to generate a relationship between P_v and L that would apply to a wide variety of landfills. The OSW database of municipal landfills served as the source for values of Q_{gen}/DC. The Scholl Canyon model for landfill gas generation (Equation 2) was used to determine the maximum expected landfill gas flowrate for each landfill in the database. In order to obtain consistency in the landfill gas generation rate between landfills, a value of 0.02 1/yr was used for k, the gas generation rate constant, and a value of 230 m^3 methane/Mg refuse was used for L_o, the gas generation potential. These values represent the 80th percentile of the k's and L_o's that were randomly assigned to the landfills in the database to obtain national and economic impacts. More information on k and L_o is provided in Chapter 3.

The resulting values of Q_{gen}/DC ranged from .000025 cfm/Mg to .0007 cfm/Mg. The average was assumed to provide a reasonable, yet conservative value for Q_{gen}/DC that could apply to a wide range of landfills. Using this value of Q_{gen}/DC, the relationship between P_v and L was obtained for three types of caps: synthetic, clay, and soil. Using cover permeabilities and thicknesses provided in Table G-1, the following equations were developed for the three cover types:

Synthetic: $P_v = 1 - (4.2 \times 10^{-7} L^2 + 4.7 \times 10^{-4} L)$

TABLE G-1. COVER PERMEABILITIES AND THICKNESSES

Cover type	Permeability (m^2)	Thickness (m)	Reference
Synthetic	1.0×10^{-18}	7.6×10^{-4}	6
Clay	5.0×10^{-15}	.61	7
Soil	1×10^{-14}	.61	8

$$\text{Clay:} \quad P_v = 1 - (4.2 \times 10^{-7} L^2 + 7.6 \times 10^{-5} L)$$

$$\text{Soil:} \quad P_v = 1 - (4.2 \times 10^{-7} L^2 + 3.8 \times 10^{-5} L)$$

These equations are illustrated in Figure 9-6 in Chapter 9.

The second step in designing an active landfill gas collection system is to determine the radius of influence that corresponds to the maximum blower vacuum determined in the first step. From Equation 5 in Section G.3, a relationship between radius of influence for an active system (R_a) and blower vacuum (P_v) can be obtained.

- Derivation of R_a as a function of Pv·

  Given Equation 5

$$\frac{P_1^2 - P_v^2}{P_v} = \frac{R_a^2 \ (\ln(R_a/r) \quad LFG \quad refuse \ Q_{gen}}{DC \quad k_{refuse} \quad (WD/L)}$$

  Solving for R_a

$$R_a^2 \ln (R_a/r) = \frac{P_1^2 - P_v^2}{P_v} \ \frac{DC}{Q_{gen}} \ \frac{k_{refuse} \ (WD/L)}{LFG \quad refuse}$$

  Using the following values:

$$r = .3048 \text{ m}$$
$$k_{refuse} = 3.743 \times 10^{-13} \text{ m}^2$$
$$WD/L = 0.75$$
$$LFG = 1.15 \times 10^{-5} \text{ N-sec/m}^2$$
$$refuse = 650 \text{ kg/m}^3$$

  the expression becomes

$$R_a^2 \ln(R_a/.3048) = \frac{P_1^2 - P_v^2}{P_v} \ \frac{DC}{Q_{gen}} \ 8.06$$

  Using the average value of Q_{gen}/DC provided in the derivation of P_v as a function L and assuming a landfill gas pressure of 1.01 atm, the expression becomes

$$R_a^2 \ln(R_a/.3048) = (1.02 - P_v^2/P_v)(1.7 \times 10^4)$$

  This equation is illustrated in Figure 9-7 in Chapter 9.

As mentioned in Chapter 9 using this approach to collection system design may result in an excessive number of wells when compared to the recommended empirical approach.

G.6.2 Simplified Approach for Passive Collection System Design

The approach outlined in Chapter 9 for passive collection systems is to determine the appropriate radius of influence for a given pressure drop across the collection and control device. The initial step in formulating this correlation was to develop a relationship between the radius of influence for a passive system (R_p) and the landfill gas pressure (P_1).

- Derivation of R_p as a function of P_1

  From Equation 17

$$\frac{Rp^2 \ln(Rp/r)}{Ra^2 \ln(Ra/r)} = \frac{[(P_1^2 - P_{atm}^2)/P_{atm}]\ Ep}{[(P_1^2 - P_v^2)/P_v]\ Ea}$$

  Assume the collection efficiencies of an active collection system and a passive collection system are equal (i.e., Ep/Ea = 1) and solve for Rp.

  From Equation 5

$$\frac{Ra^2 \ln(R/r)}{[(P_1^2 - P_v^2)/P_v]} = (DC/Q_{gen})\ \frac{K_{refuse}\ (WD/L)_{refuse}}{LFG}$$

$$\therefore\ Rp^2 \ln(Rp/r) = \frac{P_1^2 - P_{atm}^2}{P_{atm}}\ \frac{DC}{Q_{gen}}\ \frac{K_{refuse}\ (WD/L)_{refuse}}{LFG}$$

  Assuming that atmospheric pressure is equal to 1 atm and using the refuse and landfill gas properties provided in Section G.6.1, the expression becomes

$$Rp^2 \ln(Rp/.3048) = (P_1^2 - 1)(1.7 \times 10^4)$$

  To obtain the curve in Figure 9-10, the landfill gas pressure term was modified to take into account the pressure drop across the collection/control device.

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TECHNICAL REPORT DATA
(Please read Instructions on the reverse before completing)

1. REPORT NO. EPA-450/3-90-011a	2.	3. RECIPIENT'S ACCESSION NO.
4. TITLE AND SUBTITLE Air Emissions from Municipal Solid Waste Landfills--Background Information for Proposed Standards and Guidelines		5. REPORT DATE March 1991
		6. PERFORMING ORGANIZATION CODE
7. AUTHOR(S) Emission Standards Division OAQPS, OAR, U.S. EPA		8. PERFORMING ORGANIZATION REPORT NO.
9. PERFORMING ORGANIZATION NAME AND ADDRESS		10. PROGRAM ELEMENT NO.
		11. CONTRACT/GRANT NO. EPA Contract 68-02-4378
12. SPONSORING AGENCY NAME AND ADDRESS U.S. Environmental Protection Agency Office of Air Quality Planning and Standards Research Triangle Park, N.C. 2771		13. TYPE OF REPORT AND PERIOD COVERED
		14. SPONSORING AGENCY CODE

15. SUPPLEMENTARY NOTES

16. ABSTRACT

This Background Information Document was developed to support EPA's proposal for New Source Performance Standards and Emission Guidelines under Section 111 of the Clean Air Act for air emissions from municipal solid waste landfills (MSW landfills). The standards and guidelines will apply to MSW landfills sized greater than 100,000 Mg of refuse in place and emit 150 Mg/yr or more annually of non-methane organic compounds (NMOC's). This document provides information for Best Demonstrated Technology recommended for the proposal as well as emission guidelines for implementation of the regulation.

17. KEY WORDS AND DOCUMENT ANALYSIS

a. DESCRIPTORS	b. IDENTIFIERS/OPEN ENDED TERMS	c. COSATI Field/Group

18. DISTRIBUTION STATEMENT Release unlimited.	19. SECURITY CLASS *(This Report)* Unclassified	21. NO. OF PAGES
	20. SECURITY CLASS *(This page)* Unclassified	22. PRICE

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