THE EXPLOITATION OF METHANE FROM LANDFILL

THESIS
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by

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"Another damned, thick, square book!
Always scribble, scribble, scribble!
Eh! Mr Gibbon?"

William Henry, Duke of Gloucester
1743 - 1805

(Taken from the Oxford Dictionary of Quotations)

ABSTRACT

A review of literature on the subject of methane exploitation from landfill is presented in conjunction with the results of experiments concerning landfill gas extraction at the Grahamstown Landfill Site.

A description of the LFG extraction system and the utilisation of LFG at the Grahamstown Landfill Site is included.

Data concerning LFG enhancement parameters, LFG compositions and flow rates, refuse composition, LFG modelling, LFG pumping trials and the economics of LFG extraction and utilisation are presented.

The indication is that LFG can be economically extracted and utilised as a heating fuel in South Africa.

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LIST OF ABBREVIATIONS

av.	average
c	cents, 100c = 1 South African Rand
cap	capita
CFC	chlorofluorocarbon
dw	dry weight
EC	European Community
EPA	Environmental Protection Agency, United States of America
ESCOM	Electricity Supply Commission, South Africa
FID	Flame Ionisation Detector
GC	Gas Chromatography
HDPE	High Density Polyethylene
hr	hour
IR	Infrared
L.P.G.	Liquid Petroleum Gas
LEL	Lower Explosive Limit
LFG	Landfill Gas
misc.	miscellaneous
MSW	municipal solid waste
mtce	million tonne coal equivalent
mtoe	million tonne oil equivalent
pa.	per annum
P_{CO2}	carbon dioxide partial pressure
P_{H2}	hydrogen partial pressure
PVC	Polyvinyl Chloride
STP	Standard Temperature (25°C) and Pressure (1 atmosphere)
TCD	Thermal Conductivity Detector
UEL	Upper Explosive Limit
UK	United Kingdom
USA	United States of America
ww	wet weight
yr	year

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CHAPTER 1

INTRODUCTION

Landfill gas (LFG) is generated by the complex microbiological and chemical processes that prevail in anaerobically decomposing organic wastes, such as food residues and plant material. The major components of the gas are methane and carbon dioxide. The methane component of LFG is a source of energy, hence the extensive exploitation of the gas in the USA, UK and Europe.

This thesis represents the first significant study in South Africa on the subject of methane exploitation from landfill. The investigation of this renewable energy resource is examined from a South African perspective and includes the following aspects:

- The microbiological process by which LFG is generated.
- The characteristics of LFG and the monitoring of composition and flow rates.
- The enhancement of LFG production with particular reference to landfill management.
- The systems used to extract LFG.
- The utilisation of LFG with particular reference to the cost thereof.
- The theoretical and empirical assessment of LFG potential.

CHAPTER 2

LANDFILL MICROBIOLOGY:

THE PRODUCTION OF LANDFILL GAS

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1. INTRODUCTION

The organic matter present in refuse, just as any other organic matter, is susceptible to decay. Landfilled refuse can be very efficiently degraded, as landfills potentially provide ideal environments for micro-organisms involved in the degradation process. It is these micro-organisms which are responsible for the production of landfill gas (LFG), a mixture of methane and carbon dioxide and the end-product of degradation.

The process of decay is initially rapid, slowing down once the air supply is exhausted. Further degradation occurs in the absence of oxygen, and the microbes which survive in this oxygen-free environment make the important process of anaerobic digestion possible. The overall breakdown of the carbon-containing material to produce landfill gas can stoichiometrically be described as follows[6]:

$$C_nH_aO_b + (n - \frac{a}{4} - \frac{b}{2})\,H_2O \rightarrow (\frac{n}{2} - \frac{a}{8} + \frac{b}{4})\,CO_2 + (\frac{n}{2} + \frac{a}{8} - \frac{b}{4})\,CH_4 \quad \ldots\ldots (1)$$

This equation represents maximum yields and must be corrected to account for biomass formation.

Municipal refuse typically contains 60% carbohydrate (primarily cellulose), 2.5% protein and 6% lipid, the balance being comprised of non-biodegradable material[22]. Composition will obviously vary between countries and between seasons. Taking the

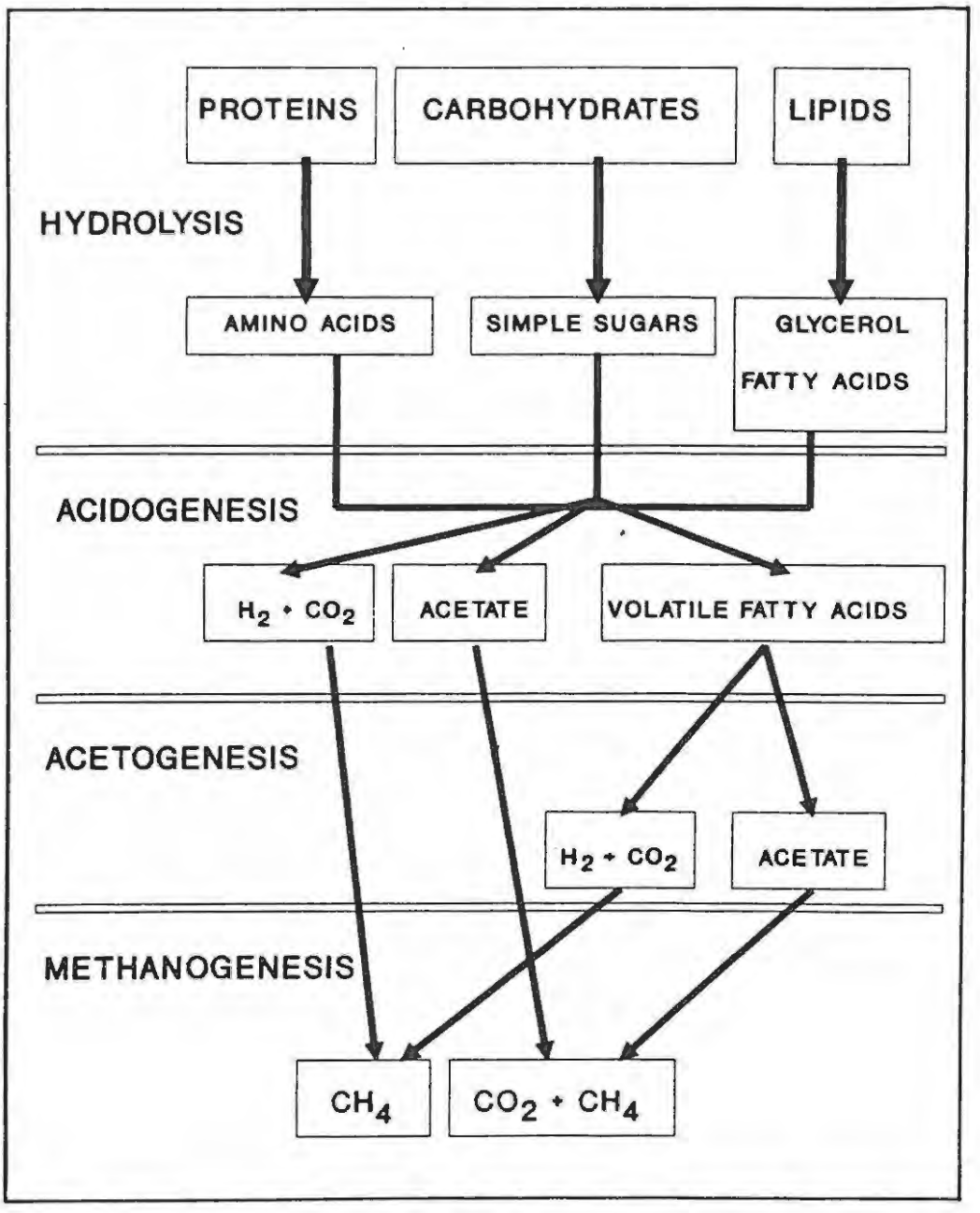

Figure 1. Anaerobic Waste Degradation Sequence.
(Adapted from Department of the Environment[8].)

organic acids with carbon chains longer than 3 (propionic, butyric, valeric and caproic), acetic acid, alcohols, hydrogen and carbon dioxide in a step known as ACIDOGENESIS.

- The production of acetic acid, together with hydrogen and carbon dioxide during ACETOGENESIS.

- The conversion of the products of acetogenesis into methane in a step known as METHANOGENESIS.

Some researchers[6,9] maintain that there are only three steps, not four, as the hydrolysis and acidogenesis stages are in fact mediated by one group of bacteria, the acidogens.

2.1 Stages of Bacterial Activity

2.1.1 Hydrolysis (Cellulolysis)

This stage is essential to the decomposition process because the solid organic material first has to be solubilised before it can serve as a substrate for bacteria. Extracellular enzymes produced by the acidogens perform this task[12]. Hydrolysis becomes a rate-limiting step in the overall degradation process if compounds which are resistant to degradation are present[8]. This may apply in the case of cellulose, which in itself is not resistant, but can inhibit the decomposition process if deposited with hemicellulose and lignin in plant-derived wastes e.g. vegetable

matter and paper. Cellulose in this form is not available to bacteria and the degradation of compounds such as paper is therefore often not complete[8].

2.1.2 Acidogenesis

The acidogenic bacteria are mostly anaerobes but aerobes and facultative anaerobes (bacteria which can grow in the presence or absence of oxygen) may be present if the landfill system is not yet depleted of oxygen, i.e. is still aerobic[11]. This group has a wide range of substrate utilisation capabilities and produces many intermediates for the anaerobic process. Examples of important reactions for this group of bacteria are given in equations (3), (4) and (5) for the fermentation of glucose[6]:

	$\Delta G°$/kJ	
$C_6H_{12}O_6 + 2H_2O \rightarrow 2CH_3COO^- + 2HCO_3^- + 4H^+ + 2H_2$ (acetate)	-206.3	(3)
$C_6H_{12}O_6 + 2H_2O \rightarrow CH_3C_2H_4COO^- + 2HCO_3^- + 3H^+ + 2H_2$ (butyrate)	-254.8	(4)
$C_6H_{12}O_6 \rightarrow 2CH_3CH_2OH + 2CO_2$ (ethanol)	-235.0	(5)

The thermodynamics for all of these fermentation reactions are favourable (i.e. the free energy, $\Delta G°$, is negative).

Hydrogen production ensures the disposal of excess electrons generated during the exothermic oxidation of organic matter during hydrolysis. This hydrogen must itself be removed, for if

it accumulates, the ensuing acidic conditions may inhibit or destroy other bacteria. While unionised organic acids, hydrogen ions and hydrogen are inhibitory to other groups of bacteria, the acidogenic population is tolerant to these intermediates[11].

2.1.3 Acetogenesis

Some researchers have rationalised that acetogenesis is the rate-limiting step in the methanogenic process because the acetogenic bacteria are the slowest growing of the trophic groups[11]. There does not appear to be agreement, however, on which group has the slowest growth rate, some stating that methanogens are the slowest growing[9]. In fact, each of the four steps in the methanogenic process have individually been termed rate-limiting by various researchers. There does not, however, appear to be any irrefutable evidence which supports the idea that any one phase is rate-limiting[17].

The acetogenic organisms utilise the products of acidogenesis, organic acids and alchols, in order to provide substrates for the methanogenic bacteria; acetate, hydrogen and carbon dioxide. Some examples of important acetogenic processes, and their corresponding ΔG° values under standard state conditions, are given in equations (6), (7) and (8)[5,6]:

These reactions cannot proceed in isolation (as indicated by the positive ΔG° values), but only when driven by other reactions for which ΔG° is negative, when the partial pressure of hydrogen (P_{H2})

is kept very low[6]. (An example of the syntrophic nature of the reactions involved in the methanogenic process is given in section 2.1.4). The reduction of hydrogen is carried out by two groups of bacteria, the autotrophic/hydrogenophilic methanogens and the sulphate-reducing bacteria[5,6,20].

	$\Delta G°$/kJ	
$CH_3CH_2COO^- + 3H_2O \rightarrow CH_3COO^- + HCO_3^- + H^+ + 3H_2$ (propionate → acetate)	+76.1	(6)
$CH_3C_2H_4COO^- + 2H_2O \rightarrow 2CH_3COO^- + H^+ + 2H_2$ (butyrate → acetate)	+48.1	(7)
$CH_3CH_2OH + H_2O \rightarrow CH_3COO^- + H^+ + 2H_2$ (ethanol → acetate)	+9.6	(8)

2.1.4 Methanogenesis

The methanogens are responsible for the terminal reactions in the digestion process, all the species producing methane as an end-product. They are very particular in their requirements, only surviving under certain specific conditions of pH and oxidation-reduction potential (redox potential, E_h) and utilising a limited range of substrates, which consists of acetate, formate, methanol, ethanol, methylamines, H_2 and CO_2[19].

The strictly anaerobic methanogenic bacteria are of two types; the hydrogenophilic/autotrophic methanogens and the acetophilic/acetoclastic methanogens. The hydrogenophilic bacteria produce methane by the reduction of formate (equation

(9)[2]) and CO_2[18,19](equation (10)[19]), while the acetophilic bacteria produce methane and carbon dioxide by the decarboxylation of acetate[14], as shown in equation (11)[6].

	$\Delta G°/kJ$	
$HCO_3^- + 4H_2 + H^+ \rightarrow CH_4 + 3H_2O$	-135.6	(9)
$CO_2 + 4H_2 \rightarrow CH_4 + 2H_2O$	-135.6	(10)
$CH_3COO^- + H_2O \rightarrow CH_4 + HCO_3^-$	-31.0	(11)

It is generally agreed that 70% of the methane produced in anaerobic digestion is due to the activities of the acetophiles[5,11,14] (equation 11), even though the reduction of carbon dioxide to methane is thermodynamically the more favoured process. The role of the hydrogenophilic bacteria, is however, of utmost importance for the removal of hydrogen from the system[18,19].

As mentioned in section 2.1.3, the anaerobic degradation of organic acids by acetogens, under standard state conditions, is energetically unfavourable. When coupled to methanogenesis, however, the degradation becomes energetically feasible, as illustrated for the case of propionate[2]:

	$\Delta G°$/kJ	
$CH_3CH_2COO^- + 3H_2O \rightarrow HCO_3^- + CH_3COO^- + H^+ + 3H_2$	+76.1	(6)
$HCO_3^- + 4H_2 + H^+ \rightarrow CH_4 + 3H_2O$	-135.6	(9)
$4CH_3CH_2COO^- + 3H_2O \rightarrow HCO_3^- + 4CH_3COO^- + 3CH_4 + H^+$	-102.4	(12)

The acetic acid produced in the coupled reaction (equation (12)) now serves as a substrate for the acetogenic bacteria, for the production of methane and carbon dioxide[2,9]. This process illustrates the close and mutual association between the acetogenic and methanogenic bacteria. Neither bacteria can exist without the other (i.e. acid degradation is energetically unfavourable, but until some degradation occurs, no substrate is available for the methanogens).

2.2 Governing Factors

The major abiotic factors affecting methane formation in landfills are oxygen, hydrogen, pH, sulphate, nutrients, inhibitors, temperature and water content, as illustrated in Figure 2[5]. Some of these variables are themselves affected by the methane generating ecosystem; levels of hydrogen and inhibiting compounds, as well as temperature and pH.

2.2.1 Oxygen

A highly reduced environment is essential for the growth and

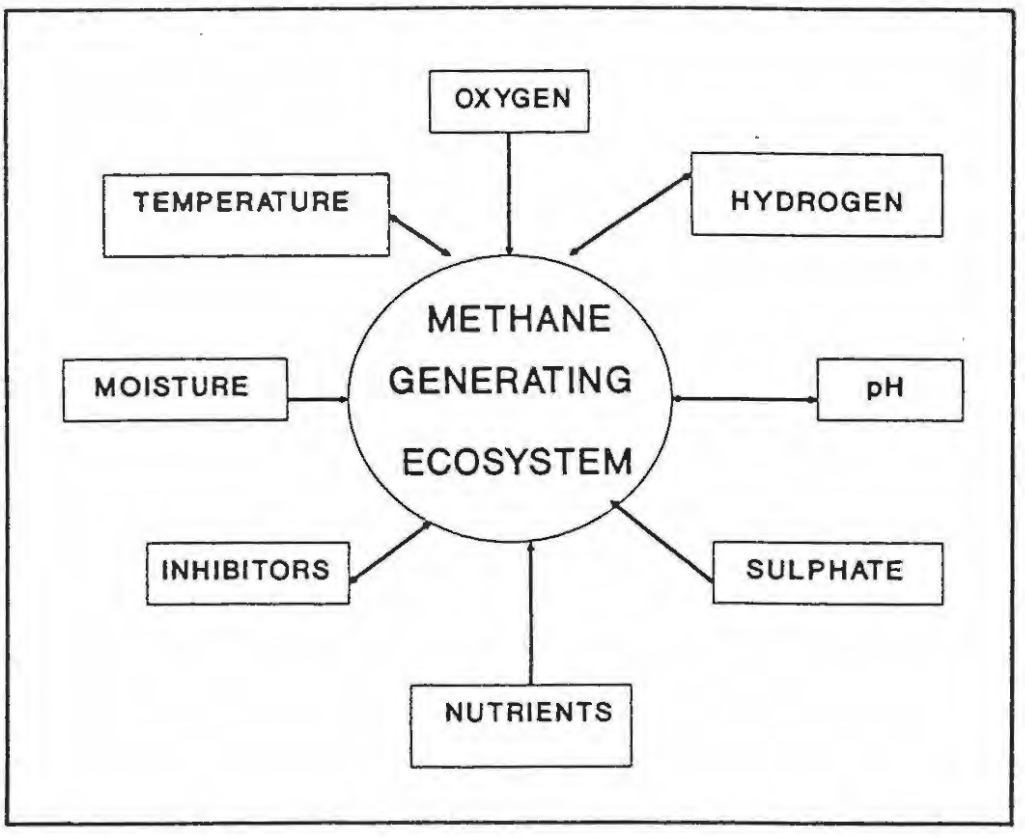

Figure 2. Major Factors Affecting the Methane Generating Ecosystem[5].

activity of the anaerobic bacteria. The redox potential (E_h) should be below -330 mV[5] although some researchers state that a potential less than -200 mV is satisfactory[10]. Above these E_h values, the bacteria become inhibited but are not necessarily destroyed.

Unless the landfill is covered with an impermeable, gas-tight cover, some oxygen will diffuse into the site from the atmosphere, especially during extensive landfill gas extraction. This oxygen is, however, rapidly consumed by aerobic bacteria in the upper layer of the landfill, so that the aerobic zone is usually of limited depth[5]. This may not be the case if pumping creates too great a vacuum, causing considerable air ingress and a subsequent deep aerobic zone in the landfill. (See Chapters 4 and 10.)

2.2.2 Hydrogen

The impact that P_{H2} has on the sequential conversion of organic matter to methane has already been stressed in sections 2.1.3 and 2.1.4. If the hydrogen-consuming bacteria, the hydrogenophilic methanogens and the sulphate-reducers, become inhibited for some reason, the low hydrogen pressure in the landfill cannot be sustained, with the result that volatile fatty acids accumulate. Propionic acid in particular tends to collect, propionic acid failure of anaerobic systems being well documented[5,22].

2.2.3 pH

The activity of all micro-organisms is affected by pH, methanogens being extremely sensitive to changes in the neutral pH range in which they like to operate. Researchers[5,11,16,22] have established that the optimal ranges for the methanogens lie between pH6 and pH8. The major controlling buffer in this range is the carbonate-bicarbonate system[11]. The acidogenic and acetogenic bacteria operate in a much wider range and are not as disturbed as the methanogens by changes in pH[5,22].

2.2.4 Sulphate

The sulphate-reducing bacteria can be of importance in the landfill environment since (i) this group resembles the methanogens in a number of ways and (ii) sulphate is a major compound of many waste types (demolition waste such as ceiling and wall boards, flyash)[5]. Whether their activities are beneficial or not to the methanogenic process appears to be related to the levels of sulphate in the waste. The organic carbon mineralised by the sulphate-reducers is converted to carbon dioxide (accompanied by hydrogen sulphide production), so that a high activity of sulphate-reducing bacteria reduces the amount of organic material available for methane production[5]. The suppression of methane formation by sulphate is therefore not due to any toxic effects of sulphate, but rather simple substrate competition. Furthermore, it has been found that hydrogen and acetate are more effectively metabolised by sulphate-reducing

bacteria than methanogens, in environments high in sulphate(23).

The positive contribution made by the sulphate-reducers to methanogenesis has been established[20], and is prevalent at low sulphate concentrations. The metabolic activity of the bacteria results in a drop in P_{H2}, stimulating the acidogenic and acetophilic methanogenic bacteria. Once all the sulphate in the waste is depleted, the hydrogenophilic methanogenic bacteria take over the role of hydrogen reduction.

2.2.5 Nutrients

The most important nutrient criterion is that the C:N ratio of the waste be between 20 and 30 to 1[17]. A well-balanced composition of minerals is also essential for the good nutrition of the anaerobic bacteria. Sufficient quantities of nitrogen and phosphorus are critical, the most common type of nutrient inhibition being due to a lack of phosphorus[2]. Micronutrients such as sulphur, calcium, magnesium, potassium, iron, zinc, copper, cobalt, molybdate and selenium are also essential to the bacteria and are typically found in landfilled waste[5]. The heterogeneity of the landfilled refuse makes the existence of nutrient-limited pockets quite possible, so that inhibition of bacteria may occur at certain points in the landfill[5].

2.2.6 Inhibitors

The inhibiting effects of oxygen, hydrogen and sulphate have

already been discussed. Further inhibitory effects are posed by substrate concentrations, carbon dioxide, salt ions, heavy metals and some specific organic compounds[5].

The inhibitory effects of volatile fatty acids in the unionised form are linked to the pH of the ecosystem and have already been discussed at length in section 2.2.3.

It has been found[5] that the removal of acetic acid by methanogens to form methane and carbon dioxide is significantly affected by the carbon dioxide partial pressure (P_{co2}), while butyric acid removal by acetogens is not. This indicates that the formation of methane from acetic acid may be inhibited by the carbon dioxide pressure in the landfill.

In anaerobic systems, an important buffer system is formed by ammonia:

$$NH_4^+ \rightleftharpoons NH_3 + H^+ \qquad (13)$$

Free ammonia, the dominant form at high pH, is toxic to the bacteria involved in methanogenesis. Wastes with a high nitrogen content therefore pose a threat of ammonia toxicity[11].

Heavy metals are required as nutrients for bacteria in trace quantities only. At higher concentrations, uncomplexed metals become highly toxic to the bacteria involved in methanogenesis. These concentrations vary, depending on the metal, as shown in

Table 1. Magnesium, for example, is highly toxic above concentrations of 1 500 $mg.l^{-1}$, while potassium is tolerable up to levels of 4 500 $mg.l^{-1}$[17]. Complexing agents such as sulphate and carbonate ions are important for the removal of metal ions[11].

TABLE 1
Concentration of Potential Toxins above which Methane Production is Inhibited
(National Academy of Sciences, cited Rivett-Carnac[17])

TOXIN	INHIBITING CONCENTRATION $mg.l^{-1}$
total ammonia	3 000
soluble sulphide	200
sodium	5 500
potassium	4 500
calcium	4 500
magnesium	1 500
nickel	500
copper	100
zinc	100
chromium	200
sulphate	5 000
cyanide	25
total volatile fatty acids	3 000
unionised volatile fatty acids	30

Certain specific compounds and industrial chemicals may also inhibit bacteria, such as phenols, chloroform and formaldehyde. Antibiotics are particularly toxic to the bacteria involved in the process of methanogenesis[17].

2.2.7 Temperature

The single most important factor controlling methanogenesis, apart from waste composition, is temperature[17]. Two distinct groups of methanogenic bacteria operate in two different temperature ranges: the mesophilic group at 30-40°C and the thermophilic group at 50-70°C[5,11,17]. Mesophilic digestion is operative in landfills and a drop in temperature to below 30°C or a rise to above 40°C will inhibit methanogenesis as methane formers are temperature sensitive[17]. The important role that temperature plays in the production of landfill gas is discussed in greater detail in Chapter 4, section 2.4.

2.2.8 Moisture Content

Moisture is essential for the activity of most micro-organisms and work by Noble *et al* (cited Westlake[22]) has shown that it has the following functions:

- as a reactant in polymer hydrolysis;
- as a transport medium for nutrients and enzymes;
- for the modification of the transformational structure of enzymes;
- for the solubilisation of metabolites;
- as an aid in exposing more substrate surface to microbial attack;
- as a control of cell turgidity.

The increase in methane production with increasing moisture content is well-documented and is discussed in detail in Chapter 4 section 2.1.

Excess moisture may have a negative impact on methanogens. If too much water is initially present in the landfill, excessive leaching of soluble sugars will lead to the production of large quantities of acidic leachate*, causing a lowering of the pH and a subsequent inhibition of the methanogens.

3. PATTERN OF WASTE DEGRADATION

Based on a knowledge of the bacterial processes occurring during anaerobic digestion and the factors which affect the processes, a sequence for waste degradation has been proposed[5], based on an initial pattern developed by Farquhar and Rovers[10]. The idealised sequence, which assumes that the waste is homogeneous, involves five distinct phases. The major changes in gas and leachate composition are illustrated in Figure 3[5]. It must be stressed that the changes are not necessarily strictly related to the degradation sequence, but represent an ideal, hypothetical situation. Leachate quality in particular does not always reflect the current status of the degradation process, but rather processes which have already occurred[13].

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*Leachate is the potentially toxic liquid produced by the infiltration of water through decomposing wastes.

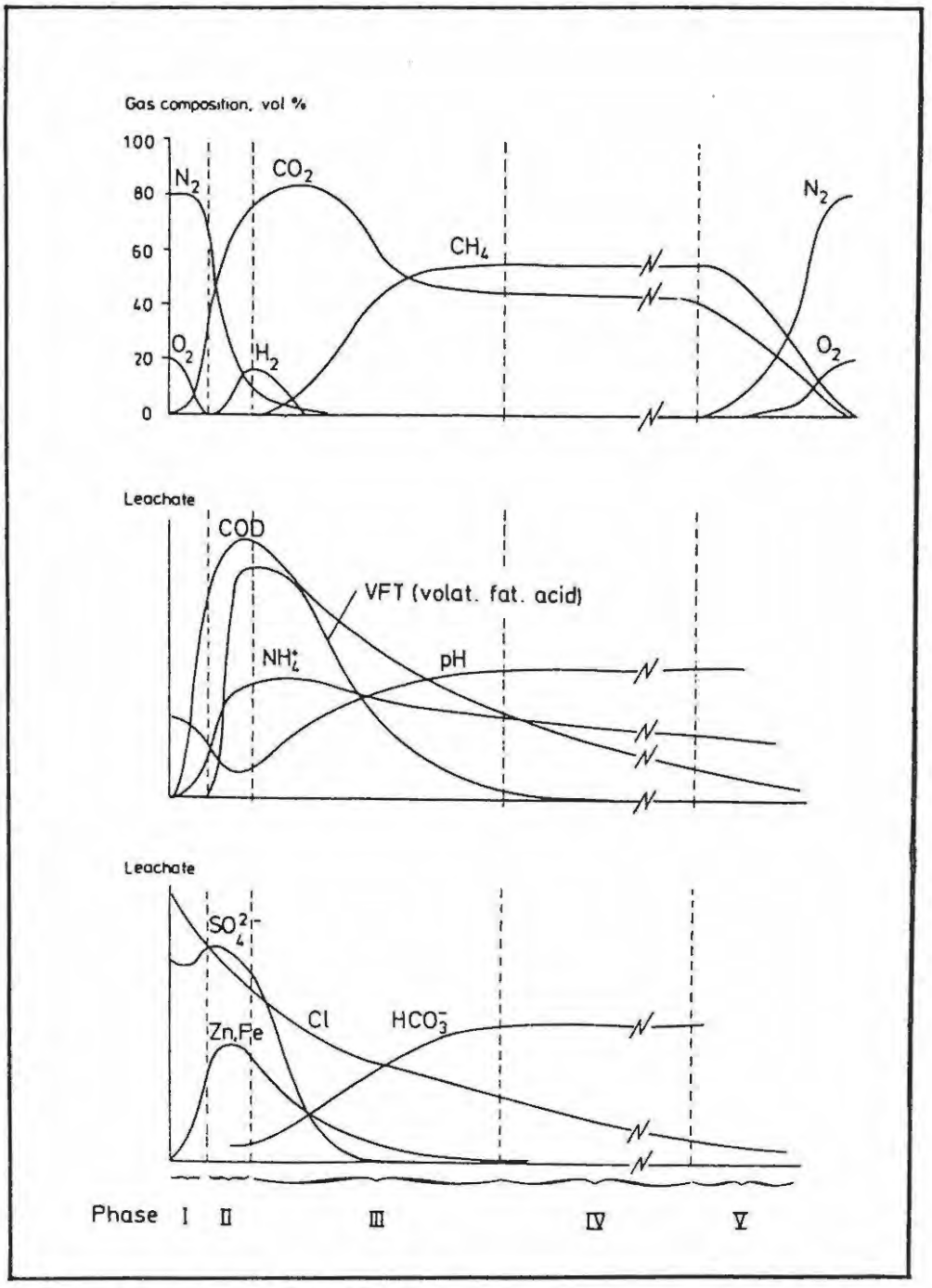

Figure 3. Gas and Leachate Composition During Refuse Degradation[5].

3.1 Phase I (Aerobic)

During this short phase (a number of days), which occurs immediately after landfilling the waste, aerobic decomposition of easily degradable organic matter takes place. The obligate and facultative aerobic bacteria produce carbon dioxide, in equivalent quantities to the oxygen they consume.

3.2 Phase II (Anaerobic Non-Methanogenic)

Upon oxygen depletion, the first intermediate anaerobic phase develops, and facultative anaerobes shift to an acidogenic metabolism. The activity of the acidogenic and acetogenic anaerobes results in the rapid generation of volatile fatty acids, ammonia (produced due to the decomposition of protein compounds), carbon dioxide and hydrogen[5,10]. The concentration of nitrogen in the gas decreases because carbon dioxide is generated in such large amounts. The increase in volatile fatty acids and hydrogen results in a decrease in the pH and the leachate generated during this phase is thus highly acidic so that heavy metals may become solubilised. Fortunately, as the redox potential drops during this period (due to oxygen depletion) sulphate is reduced to sulphide, which may precipitate the heavy metals[5].

3.3 Phase III (Anaerobic Methanogenic Unsteady)

A long phase of almost no gas production follows the previous

phase of energetic bacterial activity. Eventually the methanogens establish themselves, producing methane and carbon dioxide and consuming the acids, hydrogen and carbon dioxide produced in phase III. The pH increases to neutrality due to the consumption of volatile fatty acids, resulting in the decreased solubility of heavy metals. Sulphates continue to be produced, the produced sulphides precipitating heavy metals.[5,10].

Laboratory studies indicate that the completion time for phases I, II and III can vary between 180 and 500 days[10]. The duration of any one phase will vary between sites, being dependent on the factors mentioned in section 2.2.

3.4 Phase IV (Anaerobic Methanogenic Steady)

During this phase, gas composition and production remains stable, unless there are changes in environmental conditions. Gas compositions of 40-70% methane, and 30-50% carbon dioxide, can be expected in this phase[10]. The concentration of volatile fatty acids will continue to decrease due to the normal functioning of the methanogens[5].

3.5 Phase V (Post-Methanogenic)

In a well-managed landfill, the methanogenic phase will typically last for 10-30 years, depending on site conditions[2]. As the amount of substrate available for methanogenesis decreases, methane and carbon dioxide are produced in decreasing amounts.

Air, characterised by increasing nitrogen and oxygen contents, will begin to infiltrate the landfill. The methane present will be aerobically oxidised[4,7,15] and the waste will eventually behave as an organic peat soil undergoing mineralisation[2].

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CHAPTER 3

LANDFILL GAS

CHARACTERISTICS

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2.3 ENERGY PROPERTIES

1. COMPOSITION

The microbiological processes by which landfill gas is generated, have been discussed in detail in Chapter 2. The complex composition of this gas has, as yet, not been referred to and will be dealt with in this section.

1.1 Principal Components

Landfill gas composition varies with time, depending on the phase of refuse degradation, as discussed in the preceding chapter. During the anaerobic, methanogenic phase (the phase of interest when considering landfill gas recovery) the major components are methane and carbon dioxide. These are generally monitored as being present in the ratio of about 3:2, the gas typically consisting of 40-70% methane and 30-50% carbon dioxide by volume[4,6]*. The values reported by researchers differ between sites, which is understandable given that gas evolution is a complex process dependent on many site-specific variables.

It is sometimes reasoned that methane concentrations exceed those of carbon dioxide solely due to the solubility of carbon dioxide in water, and that were it not for carbon dioxide solubility, equivalent quantities would be monitored. This reasoning is not, however, entirely correct, and stems from the misinterpretation

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*In this thesis, STP refers to standard temperature (25°C) and pressure (1 atmosphere). Unless otherwise specified, standard state conditions are referred to throughout this report. Gas compositions are refered to on the basis of % by volume at STP.

of an equation describing the overall decomposition of organic material[4]:

$$(CH_2O)_{2n} \rightarrow nCO_2 + nCH_4 \qquad (1)$$

This equation summarises the overall reaction for an organic substrate which undergoes methanogenic decomposition. Ultimately, over the entire period required for decomposition, the end products are equivalent quantities of methane and carbon dioxide. This does not mean that equivalent quantities are produced at any particular moment. The definition of decomposition phases discussed in the preceding chapter, clearly illustrates this (Chapter 2, Figure 3). Theoretically, once the anaerobic, steady methanogenic state (Phase IV) has been reached, the gas consists of 40-70% methane and 30-50% carbon dioxide, which is independent of the solubility of CO_2. One does not expect therefore to monitor equivalent quantities of the two gases.

Landfill gas is usually saturated with water vapour, the amount of moisture depending on temperature[4]. Apart from methane and carbon dioxide, there may be considerable quantities of oxygen, hydrogen and nitrogen in the landfill gas. The quantities and proportions of these components will depend largely on the phase of landfill decomposition, as discussed in Chapter 2. Although the level of oxygen, under true anaerobic conditions in the landfill, is zero, some oxygen is normally monitored (1-5%) due to air leaks in the sampling system and/or due to excessive rates of LFG extraction[13]. When the rate at which LFG is pumped from

the site exceeds the rate at which it is produced by the methanogenic bacteria, the ingress of air into the landfill is unavoidable. (This matter is discussed further in Chapters 4 and 10). If the gas samples monitored do contain air, for either of the two reasons given above, some nitrogen, and not only oxygen, will be detected, as air contains approximately 80% nitrogen and 20% oxygen. In fact, when monitoring for the purposes of establishing the presence/absence of air ingress, it is preferable to monitor nitrogen rather than oxygen, because (i) nitrogen is present in greater quantities in air than is oxygen and (ii) oxygen may be rapidly consumed by bacteria. The detection of nitrogen is, however, fairly complex and expensive as gas chromatography with thermal conductivity detection (see sections 1.3.1 and 1.3.2) is the only available measurement technique.

The composition of LFG changes from that measured in the landfill site if it migrates from the landfill. This is because the constituents of LFG have different physical and chemical properties and may therefore migrate at different rates. In addition to this, components may participate in certain chemical reactions, such as oxidation, solubilisation and absorption[12]. The resultant gas mixture may therefore have an unusually high concentration of any one particular component. Gas migration is discussed in greater detail in Appendix 2.

Observed ranges of landfill gas compositions in the UK are given in Table 1. Gas compositions at the Grahamstown site are

discussed in Chapter 6.

TABLE 1
Landfill Gas Composition
(After Department of the Environment[5])

COMPONENT	TYPICAL VALUE % by volume	OBSERVED MAXIMUM % by volume
methane	63.8	77.1
carbon dioxide	33.6	89.3
oxygen	0.16	20.9
nitrogen	2.4	80.3
hydrogen	<0.05	21.1
carbon monoxide	<0.001	-
saturated hydrocarbons	0.005	0.074
unsaturated hydrocarbons	0.009	0.048
halogenated compounds	0.00002	0.032
hydrogen sulphide	0.00002	0.0014
organosulphur compounds	<0.00001	0.028
alcohols	<0.00001	0.127
others	0.00005	0.023

1.2 Trace Components

In addition to the major components discussed above, landfill gas contains a multitude of trace components. These include aromatic and aliphatic hydrocarbons, inert gases, halocarbons and sulphur compounds[5], a detailed listing of which is to be found in Appendix 1. The concentration of these substances is related to the types of waste deposited and the decomposition phase of the

landfilled waste.

The characteristic odour associated with landfill gas is largely due to the presence of organosulphur compounds, thiols (mercaptans) in particular[5]. Contrary to popular belief, hydrogen sulphide (H_2S), is not usually responsible for landfill odour, with concentrations rarely exceeding 10 ppm[5]. If, however, material containing high concentrations of sulphate (e.g. gypsum) is disposed of in the landfill, the quantity of H_2S produced by the sulphate reducing bacteria may be far higher. A case where H_2S exceeded 30% by volume has been reported[5].

Certain of the trace constituents contained in the water vapour saturating landfill gas, are reportedly responsible for causing extensive corrosive damage to gas recovery plants. This phenomenon is discussed in Chapter 11.

1.3 Monitoring of components

A good monitoring programme is essential for (i) the operation of an efficient gas recovery system and (ii) the establishment of potential environmental hazards. Changes in gas composition may have important implications for both (i) and (ii).

Monitoring of methane, carbon dioxide, oxygen and hydrogen sulphide can be carried out in the field using portable instruments. Confirmation of field measurements and a more detailed analysis of gas composition should be carried out

periodically by collecting gas samples and submitting them for laboratory analysis. Technology suitable for the detection of LFG components is discussed below.

1.3.1 Gas Chromatography (GC)

This is currently the most reliable means of identifying the components in a gas mixture. All of the components in landfill gas can be detected by this technique. The instruments are not, however, suitable for use in the field due to their sensitive nature. The technique can be used to (i) verify measurements taken in the field and (ii) quantify trace components such as nitrogen, carbon monoxide and volatile hydrocarbons. (GC, with thermal conductivity detection, is the only method that can be used to determine nitrogen.)

The gas sample is injected into a carrier gas stream and fractionation takes place as a consequence of the partitioning between the mobile gas phase and a stationary liquid phase, due to the different solubilities of the components in the stationary phase. Hence the individual detection of components upon leaving the column and entering the detector, which is either a Thermal Conductivity Detector or Flame Ionisation Detector (solely for the detection of organics)[17]. GC can be interfaced with mass spectrometry (MS) for the determination of hydrocarbons, other than methane, and trace gases. The technique involves extremely high resolution and is expensive but may have to be utilised if knowledge of trace components is needed. This may be the case

when gas is being purified and commercially utilised or when toxic substances are suspected of imposing environmental hazards.

1.3.2 Thermal Conductivity Detectors (TCD)

These detectors are simple, rugged, inexpensive and accurate, and are used extensively to monitor methane and carbon dioxide in landfill gas. They are employed (i) in the field, to measure the total concentration of gases, and (ii) in the laboratory, for the detection of individual components by gas chromatography. Detection is based on changes in thermal conductivity: the sensing element is an electrically heated source whose temperature at constant electrical power depends upon the conductivity of the surrounding gas[17]. Response problems in portable instruments can arise when a mixture of gases, like landfill gas, is to be monitored. Methane and carbon dioxide have different thermal conductivities (4.56 and 2.22 respectively, in units of 10^3 $W.mK^{-1}$[11,19]), and therefore affect the detection cell differently. Suppliers have addressed this problem by manufacturing instruments which are calibrated for specific mixtures of methane and carbon dioxide[5]. If, however, measurement is not carried out in the particular environment for which the instrument has been calibrated, erroneous readings are inevitable. An instrument based on TCD, the binary gas analyser, can be used for methane measurement. In this case, monitoring requires two measurements: one of the landfill gas, and a second of the gas with the carbon dioxide removed by a filter[5].

1.3.3 Flame Ionisation Detectors (FID)

These instruments typically employ a hydrogen/air flame, which serves to pyrolyse most organic compounds. The ions formed as a result of pyrolysis, are collected at electrodes, and the resulting ion current measured[17]. The technique can only be used for monitoring very low levels (parts per million) of flammable gases, as the flame may ignite explosive mixtures[5,12]. These detectors are of little use for the monitoring of landfill gas in a recovery plant as: (i) only methane concentrations below 5% by volume (the lower explosive limit (LEL) of methane in air) can be monitored, (ii) the inorganic constituents of landfill gas, namely nitrogen, oxygen, hydrogen and carbon dioxide, cannot be detected and (iii) the instruments cannot operate in an oxygen deficient environment.

1.3.4 Infrared Analysis (IR)

Specific molecules absorb infra-red energy of specific wavelengths, the absorption involving electronic transitions between energy levels in the molecule[17]. Because of its specificity, the IR technique is suitable for the analysis of complex gas mixtures. In the case of mixtures giving rise to overlapping absorption bands, computer-based calculations can be carried out in order to determine the concentration of individual species. Fortunately, this sensitive and highly specific technique is amenable to adaptation for field use. Portable infrared detectors are available for the monitoring of low or

high concentrations of methane and carbon dioxide. These instruments employ filters which only allow radiation of a specific wavelength to affect the detector[9]. Calibration of such portable equipment can be carried out by collecting samples in the field and subsequently analysing in the laboratory by GC.

Symmetrical molecules, like N_2, O_2, and H_2 are not infrared active and so cannot be detected by conventional IR techniques.

1.3.5 Gas Indicator Tubes

The system involves drawing a specific volume of gas through a tube containing a reagent which reacts with the gas to be analysed to produce a colour change. The amount of colour change indicates the gas concentration. These tubes are not suitable for accurate monitoring of landfill gas, but may serve as a crude indication for several components in landfill gas. The method may not be suitable for measurement of landfill gas concentrations when a large number of components are present, as this can produce interference effects on the detector[5].

1.3.6 Catalytic Oxidation Detectors

Instruments that utilise catalytic heat sensing elements respond to low levels of flammable gas. The instruments can only be used to measure the particular gas for which they have been calibrated[5]. Low levels (<5%) of methane can be measured by this technique but interference effects are likely when a mixture of

flammable gases is being measured, as may be the case with landfill gas; trace quantities of organics are usually present in addition to methane. Gases which are not flammable, such as oxygen and carbon dioxide, cannot be detected by catalytic oxidation. The instruments require oxygen concentrations in excess of 12% by volume in order to ensure complete oxidation of the gas[5]. As such, measurements must be carried out in a gas/air environment. This requirement is not met in a landfill gas recovery programme.

Catalytic and thermal conductivity detectors may often be incorporated in the same instrument[5].

1.3.7 Oxygen Electrochemical Cell

Oxygen meters generally employ a two electrode electrochemical cell as the means of detection. These cells consist of an electrolyte contained between a metal anode and an air cathode, to which the diffusion of oxygen is limited by a simple capillary diffusion barrier (Figure 1). Oxygen in the gas sample is reduced at the cathode, after penetrating the diffusion barrier. The rate at which this penetration and subsequent reduction occurs, is entirely dependent on the oxygen concentration in the sample. The signal current, which is proportional to the rate of oxygen consumption, is therefore a measure of oxygen concentration in the sample[10].

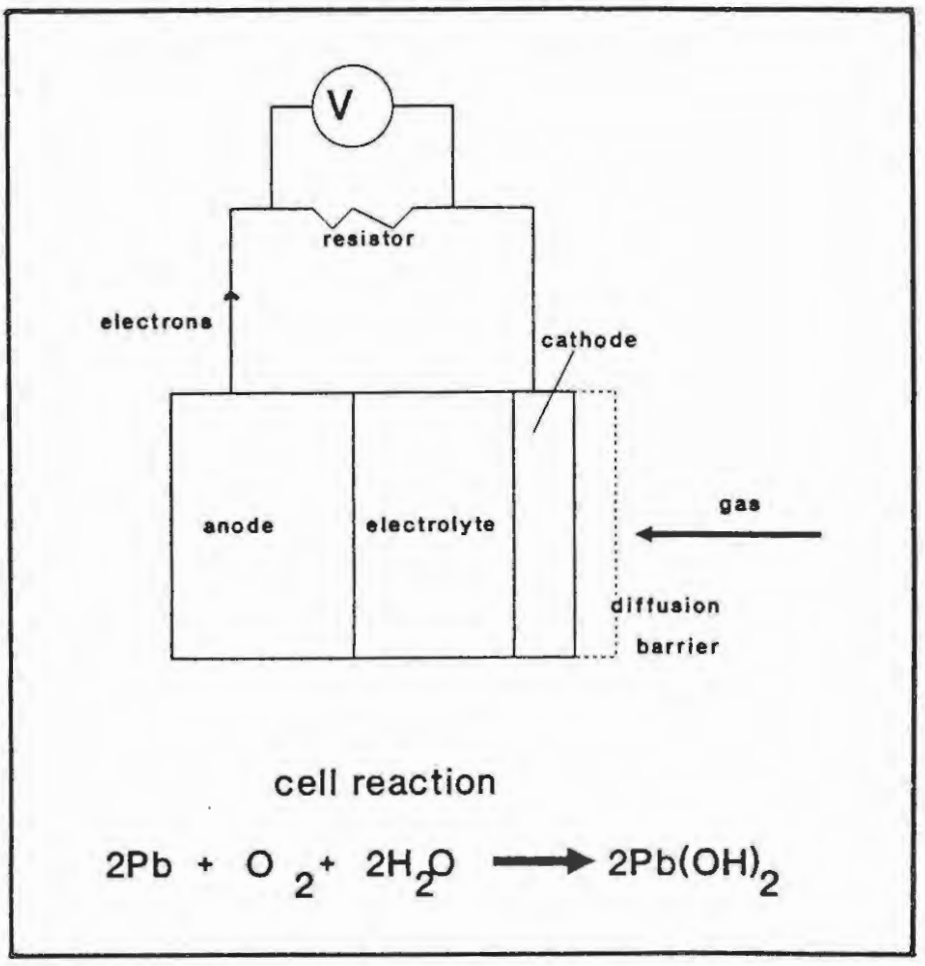

Figure 1. Oxygen Electrochemical Cell[10].

1.3.8 Chemical Absorption Detectors

An instrument employing principles of the volumetric analysis of gases, based on chemical absorption, can be used to monitor oxygen or carbon dioxide. In the case of oxygen, the gas is absorbed by chromous chloride, while for carbon dioxide, the absorbent is potassium hydroxide. The measurements are easily performed on site.

2. PROPERTIES

2.1 Fundamental Properties

Some fundamental properties of the two main constituents of landfill gas, methane and carbon dioxide, are listed in Table 2.

TABLE 2
Properties of Methane and Carbon Dioxide

PROPERTY	CH_4	CO_2	REFERENCE
molar mass /g.mol^{-1}	16.04	44.01	11
density /kg.m^{-3}	0.64	1.76	
solubility /cm^3 in 100cc cold water	_	171.3	13
lower explosion limit, LEL /%	5	_	2
upper explosion limit, UEL /%	15	_	2
normal air concentration /mg.l^{-1}	_	300	10
odour	odourless	odourless	2
colour	colourless	colourless	13

The density of methane (0.64 $kg.m^{-3}$), at STP, combined with that of carbon dioxide (1.76 $kg.m^{-3}$)[5] gives rise to a density of LFG which is between these limits, depending on the concentrations of the two gases. As a result, LFG may be heavier or lighter than air (1.15 $kg.m^{-3}$). A mixture of 60% methane and 40% carbon dioxide by volume will be slightly lighter than air, while a mixture containing 50% by volume of each component will be heavier than air.

2.2 Properties Related to Environmental Impact

2.2.1 Flammability

Methane is flammable at concentrations of 5 to 15% by volume in air, as illustrated in Figure 2. This can be explained on the basis of the reaction describing the combustion of methane:

$$CH_4 + 2\,O_2 \rightarrow CO_2 + 2H_2O \qquad (2)$$

$$\Delta H^\circ = -890 kJ.mol^{-1} \qquad \Delta G^\circ = -55 kJ.mol^{-1}$$

One mole of CH_4 ($\approx$ 25 l) exothermically reacts with 2 moles of O_2 ($\approx$ 50 l) forming carbon dioxide and water. Therefore if methane and oxygen are present in air in the volumetric ratio of 1:2, reaction (2) will proceed spontaneously. Since air comprises approximately 20% O_2 and 80% N_2, 250 l of air contains 50 l of oxygen. Hence 25 l of methane spontaneously reacts with the oxygen in 250 l of air, i.e methane spontaneously burns in

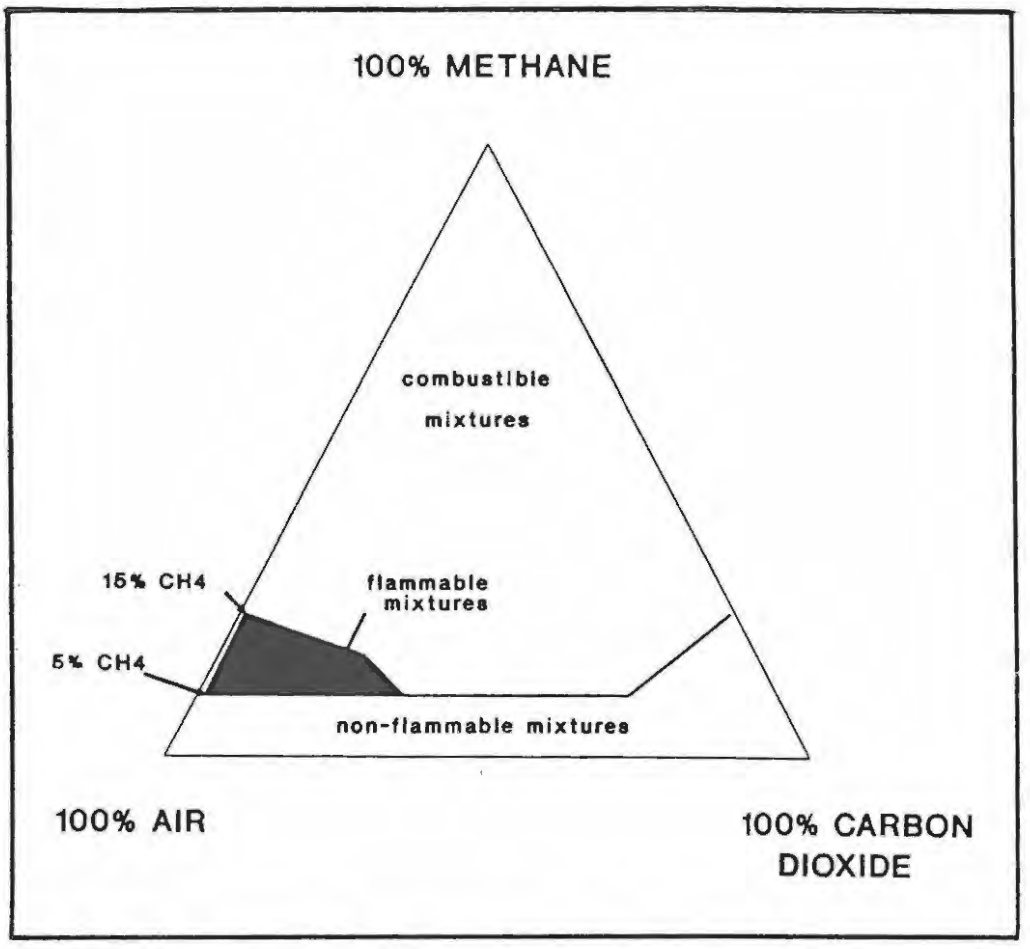

Figure 2. Methane Flammability Limits[12].

concentrations of 10% in air.

If a gas mixture which has a composition in the flammable range is contained in a closed space, in a building for example, the mixture will explode if ignited. Numerous such incidents have occurred and in the United States alone, at least 9 fatalities, from separate incidents, are documented[7]. (See also Appendix 2.)

Hydrogen, which is produced in the early stages of waste decomposition, has a flammable or explosive range in air of between 4 and 74% by volume.

The upper explosive limit (UEL) of these ranges is affected by the presence of carbon dioxide, but the lower explosive limit (LEL) is not significantly changed[5].

2.2.2 Health Hazard

In confined spaces, LFG may be an asphyxiant, as the gas is likely to contain less than 18% by volume air. Exposure to methane and carbon dioxide is known to cause nausea, dizziness and headaches. Certain of the trace constituents of LFG are toxic above certain concentrations[5,14]. Some have been classified as carcinogenic[7].

2.2.3 Odour

Methane and carbon dioxide are both odourless (Table 2). Landfill

gas, however, has a characteristic odour. This is related to trace organic sulphur compounds in the gas, as discussed in section 1.2

2.2.4 Greenhouse Effect

The greenhouse effect is caused by the accumulation of gases such as carbon dioxide, methane, nitrous oxide and chlorofluorocarbons (CFC's) which absorb or reflect the longwave radiant energy leaving the earth. The phenomenon results in a trapping of the earth's heat. There is strong scientific evidence that increasing concentrations of these gases will lead to global warming[7].

Carbon dioxide and methane, the two major constituents of LFG, are the two most important contributors to the effect (55% and 20% respectively). Methane is estimated to be 20 - 30 times more effective as a greenhouse gas than carbon dioxide, molecule for molecule[1,7,15,18]. The extent to which landfills contribute to the Greenhouse Effect has recently become an issue of priority research[18].

2.2.5 Vegetation Die-Back

LFG is documented to have phytotoxic effects on vegetation[3]. While grass can and does grow on and around landfills, deeper rooted vegetation, such as trees, does not usually survive[3], as witnessed at the Grahamstown Landfill. This may be due to (i) the displacement of oxygen from the root zone by LFG, (ii) the

toxicity of carbon dioxide to roots and (iii) the availability of heavy metals such as iron, manganese and zinc to plants, under anaerobic conditions[7].

2.3 Energy Properties

As discussed in section 2.2.1, methane burns in air as described by equation (2). The reaction is exothermic, as indicated by the negative enthalpy of combustion ($\Delta H°$), producing 890 kJ of energy. The energy in one cubic metre of methane, at STP, can be calculated using this value:

$$
\begin{aligned}
1 \text{ mole } CH_4 &= 25 \text{ dm}^3 \text{ at STP} \\
1 \text{ m}^3 \ CH_4 &= (890/0.025) \text{ kJ} \\
&= 35600 \text{ kJ} \\
&\approx 36 \text{ MJ}
\end{aligned}
$$

The energy in 1 m^3 of methane, 36 MJ, is approximately equivalent to that in 1 litre of petrol[14].

The energy available in LFG is of course less than 36 MJ, the amount of energy depending on gas composition. LFG containing 50% methane by volume will have an energy content of 18 MJ.

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CHAPTER 4

LANDFILL MANAGEMENT:

ENHANCEMENT OF LFG PRODUCTION

CONTENTS

1. INTRODUCTION

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4. SUMMARY

5. A NEW APPROACH TO LANDFILL MANAGEMENT

1. INTRODUCTION

In the past, landfilling has simply been considered as a means of getting rid of rubbish. Landfills have been perceived as a necessary evil, being associated with unsightliness, bad odour, flies and noise. The concept of a landfill as an inert dumping ground has now become obsolete and landfills are presently viewed as dynamic ecosystems, producing leachate and landfill gas as a result of the decomposition processes which characterise them. It is currently appreciated that landfills pose potentially serious environmental impacts because of the very real possibility of the migration of leachate and LFG from the site.

Leachate usually poses a greater threat to the environment than does LFG, due to its (typically) highly polluting nature during the early stages of decomposition[16]. The production of leachate, its treatment, and the largely irreversible damage it can cause to underground water sources, is well documented in the literature[4,15,16,19]. The quantity and quality of landfill leachate will not be discussed here as, although it is an integral part of landfill management, leachate has little bearing on landfill gas production. Neither the strength or the quantity of leachate necessarily reflect the current status of LFG production[29].

The adverse impacts that LFG may have on the environment have been discussed at length in Chapter 3. It is because of these threats that regulations requiring the extraction of LFG have been introduced in the USA[18], the UK and Europe.

Landfill management with the goal of minimising potential environmental impacts may be approached in one of two ways. The first approach entails treating the landfill as an inert container of waste, in which the manager attempts to keep the production of leachate and gas at a minimum. Aragno[2] is of the opinion that attempts to retard biological activity in this manner may lead to a "landfill burst". Landfills of this type may therefore be considered as potential "time bombs". In the second strategy, the landfill is perceived as a reactor or treatment device which produces leachate and gas from the decomposition process and ultimately aims to render the waste compatible with the environment at some foreseeable time in the future. This management approach is one which aims to accelerate the natural decomposition process, rather than retard or prevent it (the classical "containment" strategy), thus facilitating earlier stabilisation of the refuse[22].

Enhancement of the degradation process is desirable in order to (i) ensure that highly polluting leachate is produced as soon as, and as fast as, possible, when leachate control systems are most reliable, (ii) concentrate gas production over a shorter period of time in order to reduce gas emissions after the landfill is closed and (iii) promote earlier stabilisation of the waste, thus facilitating reclamation[40].

Numerous studies on the optimisation/enhancement/acceleration of the biological processes which take place in decomposing wastes are documented in the literature[5,6,11,12,17,21,26-28,30,31,36-38,40-43]. From

these studies it has been established that by manipulating waste degradation with the intent to accelerate decomposition, the rate of production and the methane content of landfill gas may be increased (enhanced). Good landfill management will entail that this gas is extracted and either flared or utilised, to prevent it escaping into the atmosphere or migrating underground. The benefits that enhanced gas production have for the exploitation of landfill gas as an energy source are obvious.

Researchers have investigated LFG enhancement on three scales of magnitude: (i) on a laboratory scale where sample size is typically of the order of 10-1 000 g[21,23,26,28,30,36,40], (ii) in small scale lysimeters with between 200 and 500 kg of refuse[5,6,28,42,43] and (iii) in large lysimeters or full scale landfills, where sample size is a minimum of 5 000 kg[11,12,21,31,37,38,41]. When interpreting and comparing literature results, it should be remembered that different researchers may arrive at different conclusions due to differences in test conditions, waste composition etc. The conditions under which such studies are conducted have a significant bearing on the results obtained and conditions must therefore be must be taken into consideration when comparing results.

2. ENHANCEMENT PARAMETERS

Enhancement parameters, as discussed here, are those key factors which have a favourable effect on the rate of LFG production and the ultimate yield of methane. They automatically entail the

accelerated decomposition of the landfilled refuse. They are discussed here in terms of moisture content, pH and alkalinity, nutrients, temperature, bacterial content and leachate recirculation. The parameters are examined in terms of landfill management options, with reference to their relation to the anaerobic digestion process. Some of them have already been mentioned in the discussion on landfill microbiology in Chapter 2. As will become evident, some of the parameters are interrelated and interdependent. Leachate recirculation, for example, affects both nutrient and moisture contents.

2.1 Moisture Content

Of all the enhancement parameters, moisture is probably the parameter that has been most thoroughly investigated by researchers, because it is considered to have the greatest impact on gas production[8,17,26,36,44]. Moisture is critical for the growth of bacteria and the subsequent production of LFG (see Chapter 2).

Typically, the overall moisture content of "as received" refuse is of the order of 20-30% on a wet weight basis[17], (ww)*. Once refuse is landfilled, it is exposed to the local moisture regime and may either lose or gain water, depending on site conditions.

From laboratory tests on landfill samples[23], it has been

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*Wet weight basis is referred to as (ww) and dry weight basis as (dw). The reason for including measurements on a dry weight basis is purely for purposes of comparison with field capacity values, which are sometimes given on a dry weight basis.

concluded that some methane production occurs at moisture levels as low as 10% (ww). While most researchers have established that methane generation increases with increasing moisture contents[8,9,10,17,23,26,32,36,42], others have observed that moisture addition accelerates decomposition, but not methane generation[28,43], and yet others have observed a decline in gas production[31,41]. Under conditions of rapid and excessive moisture addition, methane production rates have been observed to decrease[1,20].

The different conclusions reached by different researchers may be a function of variations in the experimental designs used. Researchers who arrived at the first conclusion, did so on the basis of laboratory scale experiments, with the exception of Rees and Grainger[32] who conducted their study at the Aveley landfill site (UK). The second finding was based on experiments in small scale lysimeters while the third has been observed in large lysimeters (>5 000 kg) at the Mountain View landfill site (California). It is evident from the literature that although enhanced methane production with increasing moisture has been theoretically hypothesised and experimentally verified on a laboratory scale, there is insufficient data from full scale landfill experiments documented in the literature to validate this premise. At present, only preliminary, and not final, conclusions concerning enhancement by means of water addition can be drawn[38].

Researchers[26,36] have established from laboratory studies that

the maximum methane production rate occurs when the field capacity of refuse is reached**.

Senior *et al*[36] observed a 2.4-fold increase in methane evolution when moisture contents were increased from 55% (ww) up to 75% (ww), the field capacity of the refuse. Above moisture contents of 60% (ww), the relationship between moisture and methane generation was not linear. Once moisture contents reached 80% (ww), a decrease in methane generation was observed. This (laboratory) observation is contrary to the (field) observation of Rees and Grainger[32] that methane generation is highest below the water table in a refuse site.

It can be argued that the percentage of methane in the LFG generated from refuse below the water table should initially increase due to solubilisation of carbon dioxide. However, considering that the pH of the system decreases due to the increased solubilisation of carbon dioxide ($CO_2 + H_2O \rightleftharpoons HCO_3^- + H^+$) and that redox potential correspondingly increases due to the increase in hydrogen ion activity (see equation (1)), methanogenesis is in fact inhibited.

$$E_h = E^\circ + (RT/zF)\ln(a_{H+z}) \quad .....(1)$$

where: E_h is redox potential/potential difference
E° is the standard potential difference
R is the gas constant, 8.314 $kJ.mol^{-1}$

**Field capacity can be defined as the incipient free moisture condition below which the movement of water is restricted and above which leachate production occurs.

T is temperature in Kelvin
z is the number of electrons
F is the Faraday constant, 96 485 $C.mol^{-1}$
a_{M+z} is ion activity

Contrary to the finding of Rees and Grainger[32], it has been observed at the Grahamstown landfill that once refuse becomes saturated with groundwater/leachate, no methane is detected in the gas extracted from this refuse (see Chapter 6). Whether this is a biochemical or purely physical phenomenon is open to debate. It may be that it is simply physically impossible to suck gas through saturated waste (see Chapter 5). If it is in fact due to biochemical effects, the author believes that apart from the inhibition of methanogenesis due to changes in pH and redox potential, inhibition may also be a result of the lowered temperature in the water-saturated ecosystem.

Some researchers[35,41] have observed that rapid and excessive infiltration of water leads to decreased methane evolution. This phenomenon was observed in a series of laboratory test cells[35], and also in large scale lysimeters at the Mountain View landfill[41]. In the first case it was attributed to an increase in redox potential, and in the second to a drop in temperature. It is somewhat surprising that the gas generation rates measured for 5 years from Mountain View, indicated that higher moisture contents lowered methane production rates[31]. Measurements at the Aveley landfill indicate increasing methane production with increasing saturation[32], as did 2 years of measurements from small scale (400 kg) lysimeters[42].

Moisture contents of 50-60% (ww)[14] and 60-80% (ww)[20] have also been established to be optimal for methanogenesis, from laboratory experiments. The different optimum moisture contents established by researchers may very well be a reflection of the different field capacities of refuse used in the studies. Documented values of field capacity range from 40% (ww)[23] to 85% (ww)[17], depending on the density/compaction of the refuse.

The degree of compaction of the refuse is a crucial factor when discussing optimal refuse moisture contents for methanogenesis. The higher the degree of compaction, the lower the permeability of the refuse and the correspondingly less moisture infiltration and vice versa. However, if compaction is too low (<0.4 tonne.m^{-3}), too much oxygen is initially present in the landfilled material and the onset of methanogenesis will be delayed. The author suggests that the optimum compaction for moisture infiltration is between 0.6 and 0.8 tonne.m^{-3}.

Although it cannot be stated so with any certainty, because of a lack of field data, it appears, from laboratory experiments, that methane generation can be enhanced by moisture addition. The indication is that the moisture contents required for optimum methane production are very site-specific and that they may vary between the limits of 40% and 80% (ww) and relate to the field capacity of the landfilled waste.

2. pH and Alkalinity

During the initial stage of degradation, refuse undergoes fermentation with the production of carbon dioxide, which upon dissolution creates acidic conditions which can lead to a lowering of the pH of the system[17,25]. This may have detrimental consequences for the methanogenic process as the mesophylic methanogenic bacteria in landfills only operate in the pH range 6 to 8[33], and are very sensitive to changes in pH (see Chapter 2).

The question arises as to whether or not the addition of an external source of buffering capacity might preclude the potential initial inhibition of methanogenesis and/or result in an earlier onset of methanogenesis. Researchers have investigated the potential enhancement of methane generation by the addition of buffers to refuse in laboratory scale experiments and lysimeters[5,6,25,28,41].

Results from laboratory and lysimeter experiments indicate that the addition of calcium carbonate ($CaCO_3$) buffer (pH≈7.5) has a beneficial effect on refuse methanogenesis[5,6,28,41]. The addition of phosphate buffer to laboratory refuse columns has, however, been found to promote acidogenesis and inhibit the methanogenic process[25].

Campbell and Croft[12] found that in test cells in the Brogborough landfill (U.K.), an increase in methane content and production

rate was observed in cells in which commercial and industrial wastes were co-disposed with domestic waste. This was considered to be a result of the buffering capacity afforded by commercial/industrial wastes.

The optimum relationships between pH, alkalinity and the percentage of CO_2 in the LFG have been established using methane digesters (Figure 1)[17]. It is seen that as the CO_2 content of the gas increases, increased alkalinity (measured as $CaCO_3$) is required to maintain a given pH. A bicarbonate alkalinity in excess of 2 000 ppm $CaCO_3$ is considered optimal[20].

2.3 Nutrients

It has already been mentioned in Chapter 2 that the most important nutrient criteria for the methanogens are that the C:N ratio be between 20 and 30 to 1[33] and that sufficient quantities of phosphorus are available. (The most common type of nutrient inhibition is that due to a lack of phosphorous[3]).

From research conducted on the effects of nutrient addition, it was found that additions of nitrogen and phosphorous in the form of ammonium phosphate ($(NH_4)_2HPO_4$), urea ($(NH_2)_2CO$) and/or potassium phosphate (K_2HPO_4) shortened the initial stage of decomposition and increased methane yields[28,30,43]. The continued addition of nutrients once methane production had started, did not improve the rate of methane production, indicating a nutrient deficiency during the initial stage of decomposition[30].

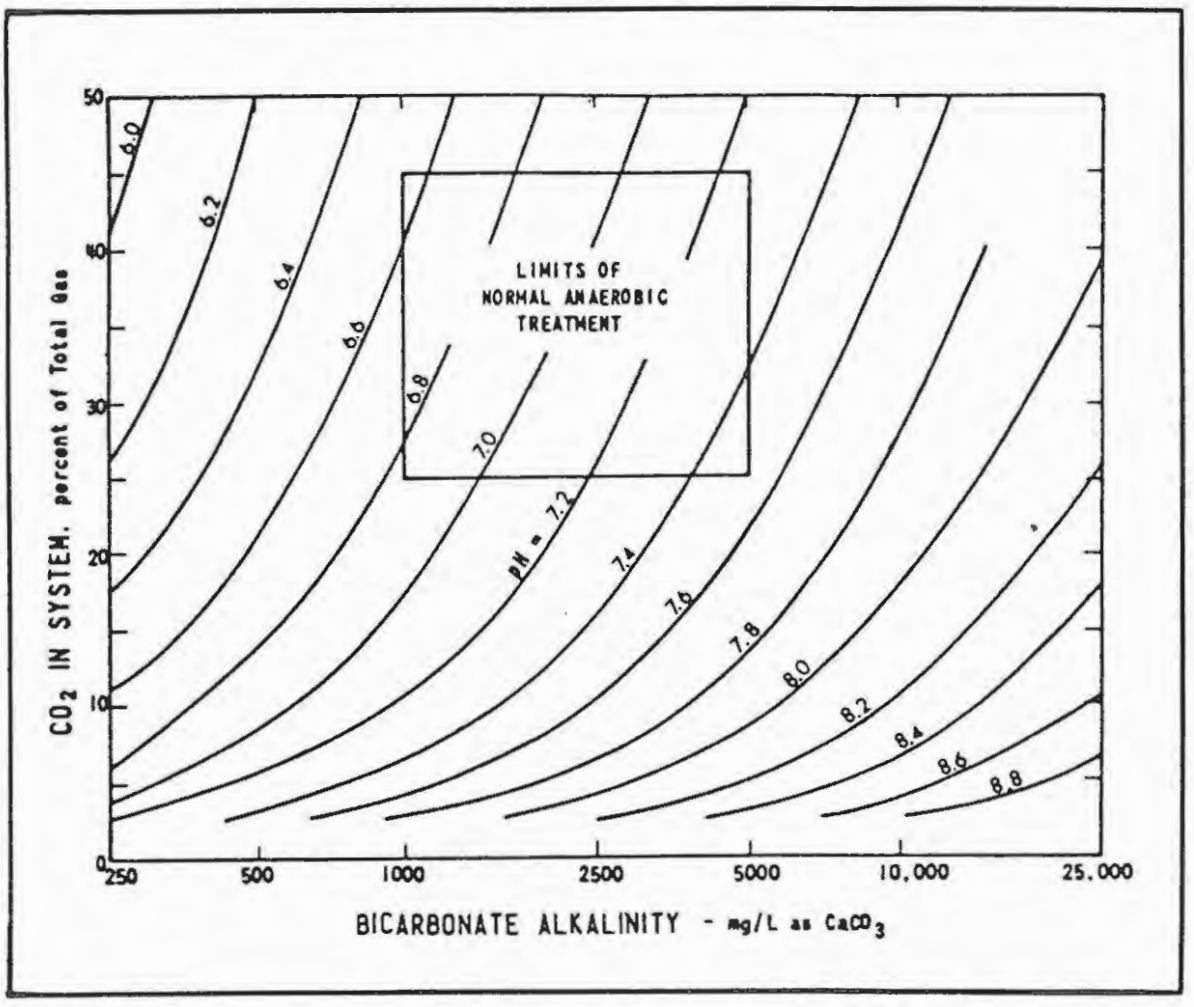

Figure 1. Relationships among Alkalinity, pH and Carbon Dioxide in Anaerobic Digestors[14]

2.4 Temperature

As discussed in Chapter 2, methanogens are operative in the mesophilic temperature range (30-40°C)[33].

Kasali and Senior[27] showed in laboratory studies that elevation of the temperature of refuse samples from an ambient temperature of 19°C to 30°C caused a 2.6-fold increase in methane production rate. An increase from ambient temperature to 40°C resulted in a 7.8-fold increase in the methane generation rate, and further elevation of the temperature to 55°C resulted in the termination of methanogenesis[27,36]. Kinman *et al*[28] observed an increase in methane production upon increasing the temperature of lysimeters to 35°C.

It must be concluded that temperature has an important role to play in the methanogenic process taking place in landfills. The optimum temperatures (35-45°C) can be attained in landfills, as evidenced by temperatures of 45°C observed in the actively methanogenic Aveley landfill[32]. Landfill practices have been developed by researchers to facilitate the achievement of temperatures of 35-45°C[32,37,38]. They entail placing an initial layer of uncompacted refuse, or aerobically stabilised refuse, at the bottom of the landfill cell. Exothermic aerobic catabolism then makes a significant contribution to the decomposition of this refuse and high temperatures (in excess of 60°C) develop. The heat generated warms refuse layers placed on top of this layer, thus ensuring that the optimum temperatures can be reached

in the anaerobically decomposing waste.

A laboratory experiment to determine the optimum temperature conditions under which methane is produced by anaerobically digesting cow dung was performed by a student under the supervision of the author. Small quantities of dung, approximately 500 g, were separately digested at temperatures of 35°C and 50°C. It was found that digestion at 35°C produced significant quantities of methane but that no methane was produced at a temperature of 50°C. The apparatus used in the experiments is schematically depicted in Figure 2.

2.5 Bacterial Content

The micro-organisms which are involved in methanogenesis come from two sources: the refuse itself and the soil into which the refuse is landfilled and with which it is covered[9,17]. Seeding of the refuse with bacteria from another source, is thought to promote a faster rate of development of the bacterial population. Accumulation of organic acids in the landfill, which inhibits methanogenesis, is thereby prevented[31]. Seeding experiments have traditionally involved the use of anaerobically digested sewage sludge and/or anaerobically degraded refuse. Both these materials are considered to be good sources of microbial inocula[30]. Experiments in which old refuse was added to laboratory scale lysimeters all showed positive effects on methane generation rates[5,6,37]. The addition of sewage sludge to refuse in laboratory scale experiments was observed to either have no effect[6] or (in

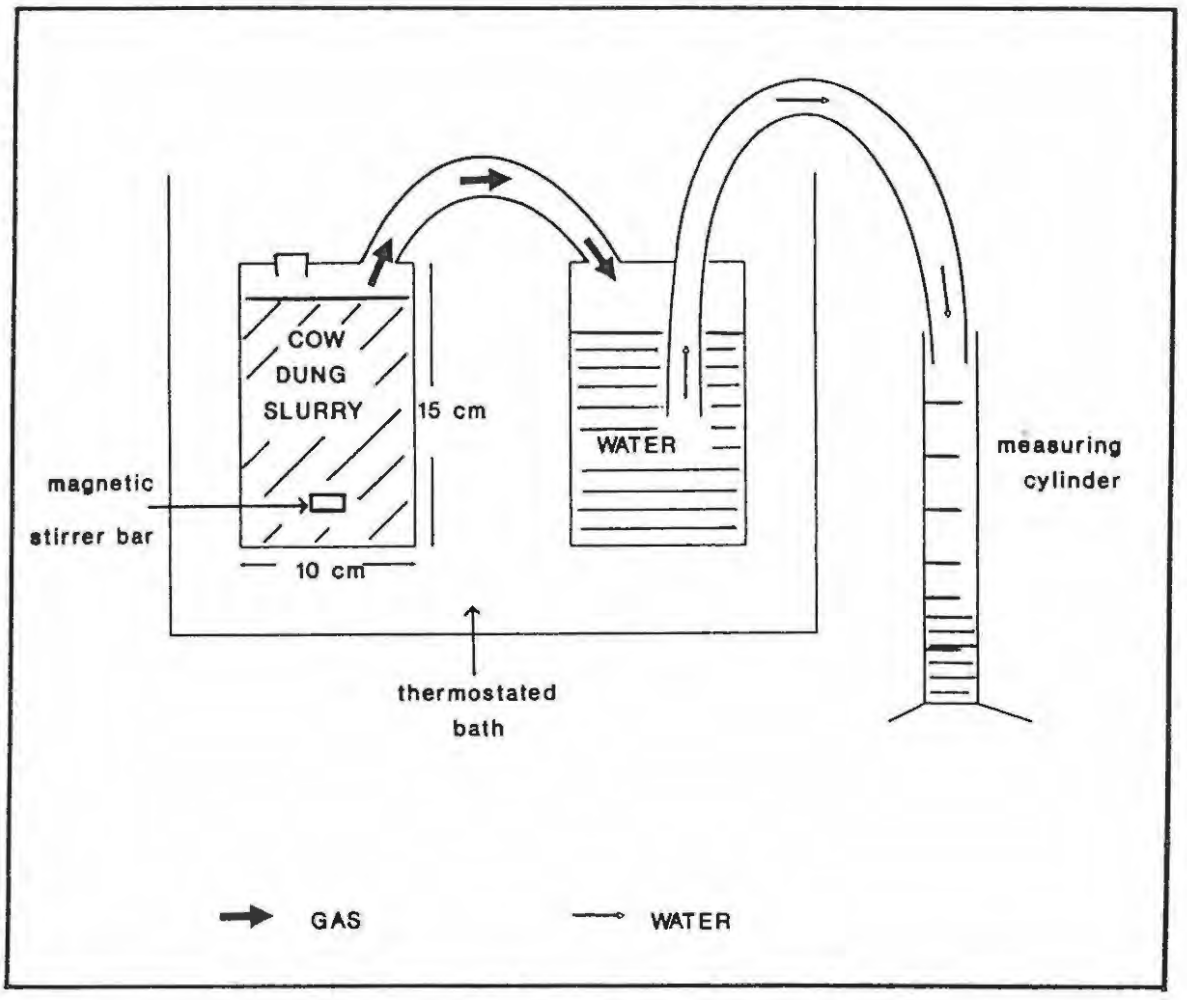

Figure 2. Apparatus used for Anaerobic Digestion of Cow Dung

most cases) was found to enhance methane production[17,21,28,30,34,43].The addition of sludge to landfill test cells has been found by Campbell and Croft[12] to give rise to increased methane production rates, although it was decided that at the time in question, it was as yet too early to make any definite conclusions. Pacey[31], on the other hand, measured decreased methane generation rates once sludge had been added to lysimeters in the Mountain View landfill.

The positive effects that have been observed from the addition of sewage sludge are not necessarily related to the addition of a microbial inoculum[38]. It is quite possible that the sludge simply serves to increase the moisture and nutrient contents of the refuse[12].

Most researchers are of the opinion that sewage sludge addition leads to enhancement[17,21,28,30,34,43]. Stegmann and Spendlin[38], however, believe that no definite conclusions can be drawn as in experiments performed to date, different kinds and amounts of sludge were utilised, thus making the comparison of results difficult.

2.6 Leachate Recycle

Leachate recirculation may facilitate the transport of moisture, nutrients and bacteria through the landfilled waste and thus enhance methane production[28,31]. Leachate treatment is costly and the practice of recirculation thus has the added benefit of

providing a means of treating the leachate in the landfill itself[40].

Leachate recirculation in lysimeter and laboratory experiments is performed with or without the addition of nutrients and/or buffer to the leachate.

Barlaz *et al*[5] and Walsh *et al*[42] observed enhanced methane production when leachate recycle, with no additions, was practised in laboratory scale lysimeters. Other researchers[43] could only conclude that leachate recycle appears to accelerate decomposition. Leuschner[30] observed in laboratory experiments that recycled leachate, which had been buffered and supplemented with nutrients, enhanced methane production. Leachate recycle without addition, however, was found to be ineffective in enhancing methane production, and produced a "sour" digester. Kinman *et al*[28] observed that leachate recycle, without addition, accelerated decomposition in laboratory scale lysimeters for the first two years, but that gas production declined rapidly thereafter. It was concluded that the practice of leachate recycle had produced toxic effects in the methanogens.

Stegmann and Spendlin[38] are of the opinion that controlled leachate recirculation at full scale landfills may have beneficial effects if mean annual precipitation is less than 800 mm.

3. MANAGEMENT OF THE GRAHAMSTOWN LANDFILL SITE

The site for the Grahamstown Municipal Landfill was chosen on the basis that it posed negligible potential environmental impact.

The landfill is situated in a disused kaolin clay mine, with an available space of 10×10^6 m^3 which still has vast reserves of clay (see Photograph 1 and Figure 3). As such, it is considered to be an ideal site from a hydrogeological perspective. The possibility of groundwater contamination by leachate is minimal, due to the absorptive capacity and fairly low permeability of the huge clay reserves.

Until recently, landfills were always sited in areas deemed worthless for any other purpose, such as large scale excavations. It is now maintained that they should be constructed above ground, with a safe distance to the groundwater surface, in order to prevent contamination of groundwater[39]. The Grahamstown landfill, even though it is situated in a pit, is however, considered to be "safe" because of its natural clay liner.

3.1 Landfilling Procedure

Refuse has been landfilled since 1986, according to the well-known cell method, in which pre-constructed clay-walled areas form cells which are filled with refuse. The cells have an approximate volume of 150 m^3 and are contained by 0.5 - 3 m thick clay walls, the thickness being determined by whether or not the

Photograph 1. Aerial View of the Grahamstown Landfill, 1989.

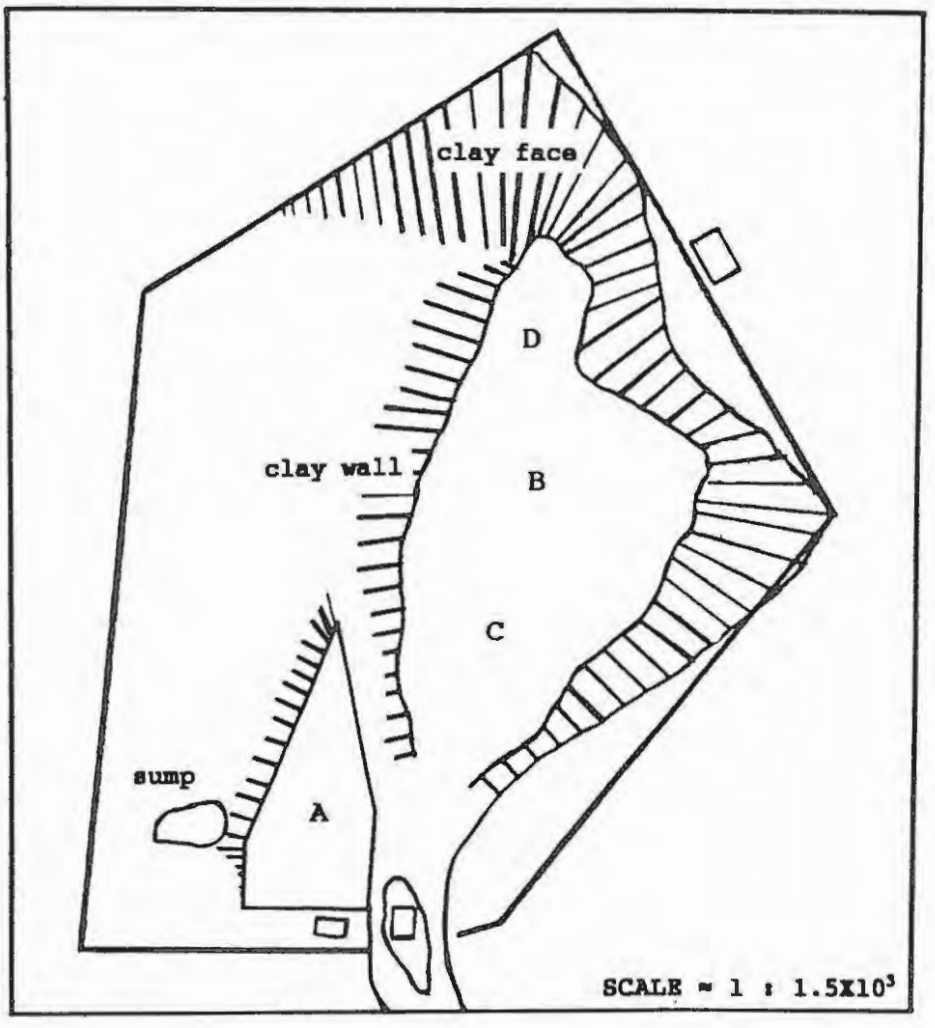

Figure 3. Schematic Diagram of Grahamstown Landfill Site

A = rehabilitated area
B = upper tipping area
C = lower tipping area
D = upmost tipping area

cell is on the edge of the site. Domestic refuse, which arrives on site in a compacted form, is mixed with other MSW (garden refuse, builders rubble etc.) and compacted (≈ 0.7 tonne.m^{-3}) by a front end loader in the cell presently being worked. The refuse is then covered with a thin layer (<50 mm) of the soil that is disposed of as builders rubble or, if this is not available, with clay from the site. The practice of covering and compacting has the effect of (i) conserving space, (ii) controlling odour and flies, (iii) preventing wind-blown plastic and paper and (iv) minimising the amount of oxygen in the refuse, which promotes the development of anaerobic conditions and reduces the fire hazard.

The landfill site is schematically depicted in Figure 3. Refuse in section A of the figure, amounts to 5 000 tonnes and occupies a volume of 6 500 m^3. This part of the site, which is 10 m deep, has not received refuse since 1988. Upon termination of tipping activities in this area the site was capped with a 40 - 60 cm layer of clay, overlain by 10 - 20 cm of soil on which grass has been planted (see Photograph 2).

At present, refuse is deposited in the sections labelled B, C and D, which together contain some 55 000 tonne of waste and occupy a volume of 80 000 m^3 (see Chapter 8). Refuse depth in this area varies between 10 and 20 m.

A leachate sump is located at the lowest point in the site, just below the rehabilitated section. The leachate generated by the decomposing waste therefore collects in this sump, from where it

Photograph 2. The Rehabilitated Site.

is pumped to the top of the rehabilitated section.

3.2 Investigation of Enhancement Parameters

3.2.1 Moisture

In order to determine the moisture status of the Grahamstown Landfill, numerous moisture measurements have been carried out. These include (i) determination of the moisture content of refuse samples by oven drying (ii) determination of the field capacity of refuse samples and (iii) determination of a moisture profile in the site.

Moisture Content and Field Capacity

Random samples of landfilled refuse (≈ 1 000 g) were obtained from the rehabilitated portion of the landfill by one of two methods. In Winter 1990 and 1991, samples were extracted from a depth of 80 cm with a hand auger. During Summer 1990/91 they were taken at a depth of 100 cm from a pit which had been dug into the side of the refuse for the *in situ* installation of psychrometers (section 3.1.2). The samples were oven dried at 105°C for 5 days (by which time constant weight had been reached) and the moisture contents calculated on a wet weight (total weight) basis.*** Averaged results are given in Table 1.

***The calculation of moisture content on a wet weight basis is as follows:

$$\text{moisture}/\% = \frac{\text{wet-as-received sample weight - dried sample weight}}{\text{wet-as-received sample weight}}$$

The samples obtained in Summer 1990/91 were also analysed by the Civil Engineering Department at the University of the Witwatersrand. These analyses were carried out in order to establish the relationship between moisture content and field capacity in the refuse samples. It is recognised by many researchers that the optimum moisture conditions for methane generation occur at field capacity[36] (see section 2.1). Moisture contents were determined on a dry weight basis**** once samples reached constant weight while drying at 50°C. The author has converted these values to moisture contents on a wet weight basis for reasons of comparison with the values obtained by the author (see Footnote ****). Field capacity was also determined on a dry weight basis (the only reason for including the values of moisture content on a dry weight basis is for reasons of comparison with field capacity values). Results are given in Table 2.

****The calculation of moisture content on a dry weight basis is, by convention, as follows:

$$\text{moisture/\%} = \frac{\text{wet-as-received sample weight - dried sample weight}}{\text{dried sample weight}}$$

OR

$$\frac{\text{weight water}}{\text{dried sample weight}}$$

The value so obtained is referred to as the % moisture content on a dry weight basis.
Moisture contents on a wet weight basis can easily be determined from moisture contents given on a dry weight basis. For example, if moisture content on dry weight basis = 65%:

$$65\% = \frac{65}{100} = \frac{\text{weight water}}{\text{dried sample weight}}$$

Then moisture on wet weight basis $= \frac{\text{weight water}}{\text{dried sample + water weight}}$

$$= \frac{65}{100+65} = 37\%$$

TABLE 1
Refuse Moisture Contents as Determined by the Author

SEASON	MOISTURE % wet weight basis
winter 1990	42
summer 1990/91†	27
winter 1991	40

NOTE: † These refuse samples were also sent to the University of the Witwatersrand for analysis.

TABLE 2
Averaged Results of Analyses of Refuse Samples for Moisture Content, Field Capacity and Refuse Density by the University of the Witwatersrand

MOISTURE % dry weight basis	MOISTURE† % wet weight basis	FIELD CAPACITY dry weight basis	DENSITY $kg.m^{-3}$
65	37	145	488

NOTE: † Determined by the author from the dry weight value (see Footnote ****).

It is apparent that the water contents measured by Wits University are very different from those measured by the author. For the same sample (Summer 1990/91) average moisture contents of 37% and 27% (ww) were measured respectively. It is not known why the measurements should be so different.

The moisture contents in both Tables 1 and 2 are lower than the levels which have been established by most researchers[14,20,36] for optimum methane production. Results in Table 1 indicate that refuse moisture is 32 to 36% higher in winter than in summer. The measurements of field capacity in Table 2 show that field

capacity has not been reached in the refuse and that moisture contents (on a dry weight basis) are only 45% of the field capacity.

It can be concluded that the low moisture levels in the rehabilitated portion of the Grahamstown Landfill are possibly inhibiting the methanogenic process, as reflected by the low LFG flow rates and only moderate methane concentrations that have been measured (Chapters 7 and 8).

Landfill Moisture Profile

In order to obtain a refuse profile (and hence a moisture profile) a pit was dug into the side of the landfill. The pit was 150 cm deep and extended some 100 cm into the refuse. Three Wescor PST 55 psychrometers were placed in perforated piping, for protection, and inserted at different depths in the profile (Figure 4 and Photograph 3). The pit was backfilled with the excavated refuse with the psychrometers' cables extending from their positions in the refuse to above the landfill surface. Water potential readings on a Wescor HR 33T microvoltmeter were taken at various times, and the results plotted. The theory of thermocouple psychrometry is discussed in Appendix 3.

Before installing the psychrometers, the Wescor calibrations were checked. The calibration procedure outlined in the instruction manual was followed. A microvolt recorder output was not

Photograph 3. Installation of Psychrometers.

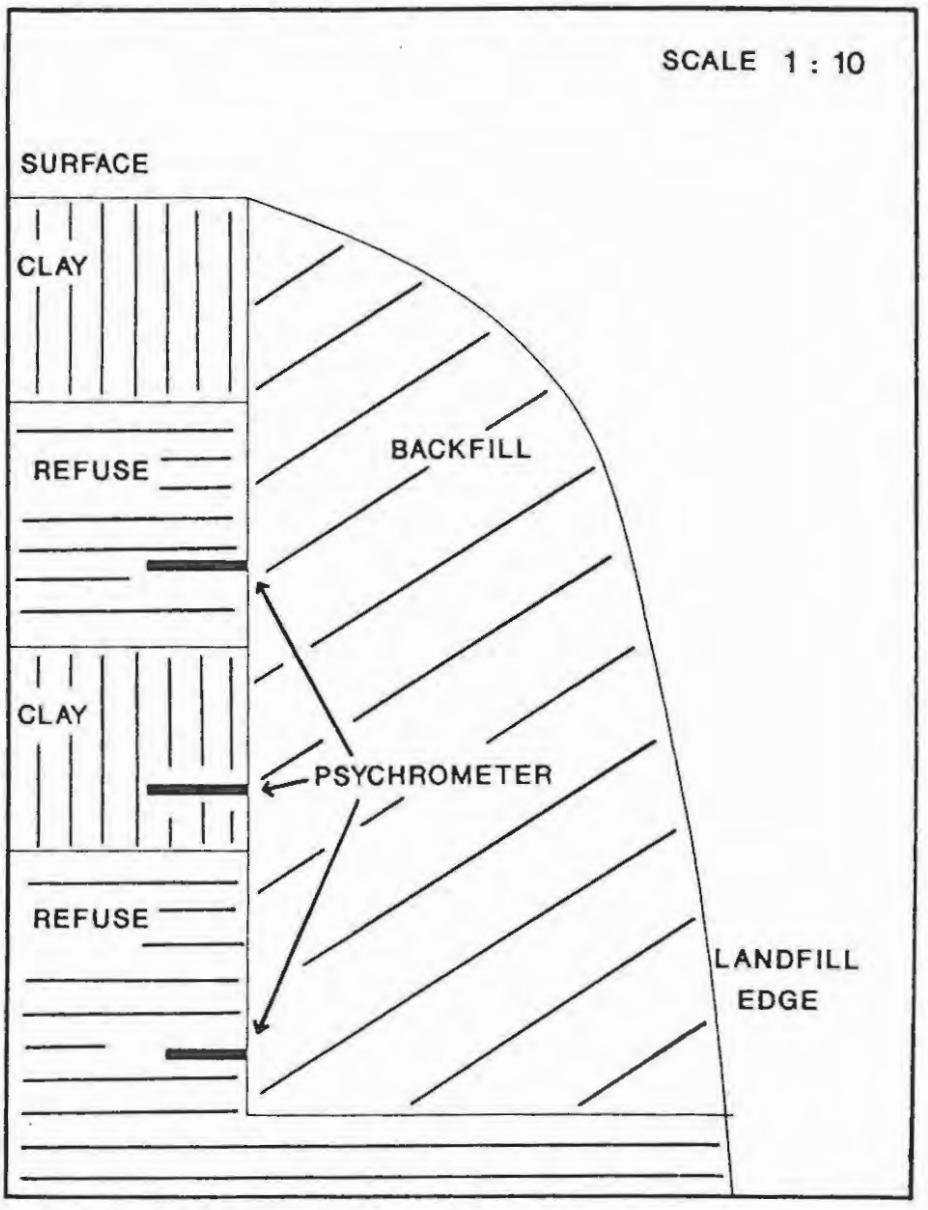

Figure 4. Psychrometer Positions in the Landfill.

obtained, but the Wescor calibrations (-25 bar***** in 0.5514 molal NaCl) were confirmed.

Water potential and temperature results are tabulated in Table 3. Each measurement was done three times and readings were in fact found to remain constant. The water potential values were obtained by dividing the microvoltmeter readings by 0.47 microvolts.bar^{-1}, this being the figure given by Wescor for the typical responsivity of thermocouple psychrometers. In order to relate all data to one temperature (25°C) the following correction procedure was used (see Wescor Instruction Manual):

$$corrected\ reading = reading\ /\ (0.325+0.027T)$$

where: T is in degrees Celsius

The readings taken on different days have been plotted and a refuse moisture profile thus obtained (Figure 5).

No readings subsequent to those taken in December and January were made, the psychrometer at the 70 cm level giving no response in the February month. The pit was dug up once this was discovered and the psychrometer found to be contaminated.

*****1 bar = 10^5 Pa

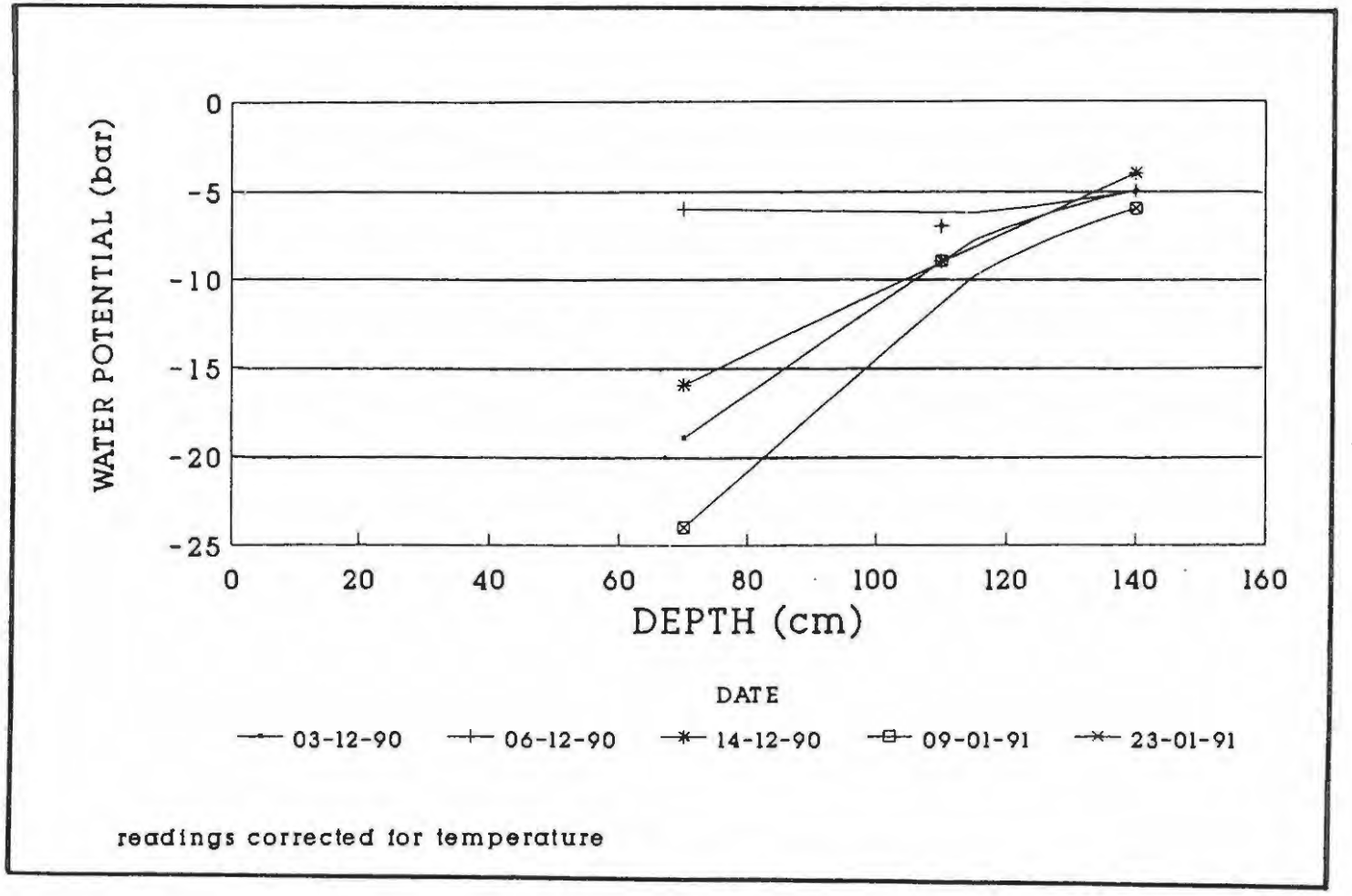

Figure 5. Refuse Moisture Profile

TABLE 3
Psychrometer Readings at Various Depths in the Landfill Site

DATE	WATER POTENTIAL /-bar			TEMPERATURE / °C		
	70cm	110cm	140cm	70cm	110cm	140cm
03-12-90	19	7	5	27	24	24
06-12-90	6	7	5	28	25	25
14-12-90	16	9	4	29	25	25
09-01-91	24	9	6	31	28	28
23-01-91	24	9	6	33	28	28

The refuse moisture profile (Figure 5) shows an increase in moisture (increasingly less negative water potential) with depth. The changes in the profile with time are an indication of the changing moisture conditions in the refuse ecosystem. The water potential near the surface (70 cm depth) varies considerably between the limits of -6 and -24 bar. The low value of -6 bar recorded on the 6 December is not surprising as a considerable amount of rain fell on the 5 December. The higher water potential values (and the moisture contents) recorded during January 1991 possibly reflect the hotter weather, and hence drier conditions, during this month. Water potentials at the 110 cm and 140 cm levels were relatively constant throughout the measuring period and the low values suggest conditions approaching saturation. In fact, while the pit was being dug, water was observed at a depth of 150 cm.

3.2.2 Leachate Recycle

Leachate recycle is practised at the Grahamstown Landfill in

times when sufficient quantities of leachate have collected in the leachate sump. Very little leachate is produced during the dry summer months so that leachate is not usually recycled during the summer season. Low refuse moisture contents are observed during this period (section 3.1).

Leachate recycling probably has a greater effect in terms of moisture rather than substrate addition, as the organic content of the leachate (measured in terms of COD, the chemical oxygen demand), is exceptionally low, 400 $mg.l^{-1}$. Analyses of the leachate in terms of pH, COD, NH_3, Cl and Fe, were carried out at bi-monthly intervals and concentrations found to exhibit little fluctuation. The averaged measurements are given in Table 4.

TABLE 4
Leachate Composition

PARAMETER	VALUE
pH	8.0
COD/$mg.l^{-1}$	400
conductivity/$mS.m^{-1}$	390
NH_3/$mg.l^{-1}$	80
Cl/$mg.l^{-1}$	750
Fe/$mg.l^{-1}$	2

The practice of leachate recycle has not appeared to enhance the methanogenic process in the rehabilitated portion of the landfill as gas flow rates remain consistently low. Far more data needs to be collected before any definite conclusions can be made.

from the extraction wells elsewhere in the site. This observation may be related to factors other than the abattoir waste, for example the young age of the refuse.

3.2.4 Containment of LFG

It has been established that while enhancement of the anaerobic process increases the rate at which LFG can be extracted from a site and possibly the methane content of the gas, extraction itself enhances methane production. This is related to the high partial pressures of carbon dioxide which can develop in a landfill from which gas is not extracted which can inhibit methanogenesis[24].

Efficient extraction relies on preventing the intrusion of atmospheric oxygen into the waste[13], which degrades the energy content of the extracted gas and causes aerobic conditions to develop. The containment of LFG (and hence the prevention of pathways for the intrusion of oxygen) can hence be considered to be an enhancement parameter. All the enhancement parameters discussed thus far will have no effects if containment of the gas is not optimised.

Experiments have been carried out on the rehabilitated portion of the Grahamstown site to determine to what extent the LFG is contained, and to determine how important containment is for the generation and extraction of LFG. As already mentioned, this portion of the site is covered with a clay cap 40 - 50 cm thick

overlain by a layer of soil which is grassed. The refuse is contained by 3 m thick clay walls on the edge of the site.

A hollow, perforated steel probe, 3 cm in diameter, was driven into the top 40 cm of the refuse and the gas in the probe monitored. The methane content was found to be 4%, suggesting that methane (and hence atmospheric oxygen) possibly can and does move through the landfill surface. A similar probe was inserted 40 cm into the clay wall on the site boundary (Photograph 4), and a methane content of 5% detected, indicating possible diffusion through the sides of the site. A further two probes were inserted into the soil at distances of 1 and 2 m from the site boundary. The probe at a distance of 1 m was monitored to contain 5% methane while that at 2 m did not contain any detectable methane. It can be concluded that some migration of methane from the site does occur. More measurements are however required to establish the extent of this phenomenon. Further information regarding gas migration, including details of an investigation at the Coastal Park Landfill, Cape Town, carried out by the author, are documented in Appendix 2.

In order to further investigate the implications of these findings, i.e that LFG containment is inadequate, the pressure in one of the probes was monitored while extracting gas (at a pressure of 2 cm water) from the gas extraction well a distance of 2 m from the probe (Photograph 5). A negative pressure of 5 mm water was observed in the manometer attached to the top of the probe. It was thus established that pathways for the ingress

Photograph 4. Installing a Probe for LFG Measurements in the Side of the Site.

Photograph 5. Measuring the Suction in a Surface Probe while Pumping on Well A2.

of air into the site do exist.

These experiments were carried out in the dry summer months of 1991, when the beginnings of cracking of the landfill cover were observed. To establish whether or not air ingress is confined to the periods when these dry conditions (and hence surface cracks) prevail, the gas composition from the extraction wells was monitored both shortly before and immediately after the first winter rainfall (June). Gas was extracted, at a pressure of 2 cm water, for a period of 20 minutes from each well. The concentrations of CH_4 and O_2 were measured after the first 5 minutes and again after 20 minutes, in order to ascertain whether or not methane and oxygen contents decreased/increased during this period (thus indicating the presence/absence of air ingress). CO_2 concentrations were only read after 20 minutes.

The results of gas extraction from well A2 are given in Table 5. For the two measurements, before and after rain, CH_4 and CO_2 comprise 57% and 76% of the LFG respectively, after pumping for 20 minutes. The trace components in LFG cannot possibly make up the rest of the volume, but only a small fraction of it, so that it can be deduced that the rest of the gas is largely made up of air. Now, considering that air contains 80% N_2 and 20% O_2 one would expect the LFG in the first measurement (before rain) to contain 30% N_2 and 9% O_2. However, only 1% oxygen was detected. This is because oxygen is readily consumed by bacteria as it is drawn through the refuse, and hence is not detected once LFG enters the instrument. Nitrogen is unaffected and hence makes up

the remainder of the gas. Similarly, the LFG in the second measurement (after rain) should contain 18% N_2 and 5% O_2, if the oxygen is not consumed by the bacteria.

It is observed in Table 5 that methane contents do not decrease and oxygen contents do not increase to as great an extent after as compared to before the occurrence of rain. This confirms that the extent of air ingress is influenced by the degree of saturation of the cover material.

In conclusion, during dry times of the year, LFG is inadequately contained in the rehabilitated portion of the Grahamstown Landfill, due to increased porosity of the clay surface. Only during the rainfall season can optimum conditions therefore be hoped to be realised.

TABLE 5
Monitoring of well A2 Before and After Rain

	CH_4 % by volume	O_2 % by volume	CO_2 % by volume
before rain	42 → 32	0 → 1.0	25
after rain	43 → 41	0 → 0	35

NOTE: For methane and oxygen concentrations, both the initial and final (after 20 minutes) measurements were recorded. For carbon dioxide, only the final measurement was taken.

3.2.5 Temperature

Temperatures in the Grahamstown Landfill Site were measured by lowering a thermocouple a distance of 5 m into the extraction

wells. Unfortunately, the thermocouple did not give very stable and reliable readings. At best it can be stated that for the rehabilitated site, temperatures varied between a minimum of 25°C and a maximum of 35°C. For the wells in the present tipping site, temperatures were substantially higher, between 30°C and 40°C. The indication is that temperatures in the present tipping area are ideal for methanogenesis. This is supported by the methane concentrations and LFG flow rates measured in this area (see Chapter 6 and 7). In the rehabilitated site, temperatures are generally on the low side for optimum bacterial activity. Active methanogenesis does not appear to be prevalent in this site as the methane concentrations and flow rates measured are low.

4. SUMMARY

The conclusions arrived at from a review of available literature and from observations made at the Grahamstown landfill, can be summarised as follows:

(1) Refuse moisture content does affect methanogenesis. Optimum moisture levels are between 40 and 80% (ww) and appear to be related to refuse field capacity. At the Grahamstown landfill, moisture is possibly the limiting factor in the methanogenic process, as moisture contents (65% (dw)) are significantly lower than field capacity (145% (dw))

(2) The addition of $CaCO_3$ to the decomposing refuse provides a good source of buffering capacity and enhances methane production.

(3) The initial lag in methanogenesis can be shortened by

additions of nitrogen and phosphorous.

(4) Temperatures for optimum methane generation, 35-45°C, can be generated in landfills by tipping the refuse quickly over a few days and then sealing off the cell in order to contain the heat. Allowance for aerobic decomposition of a bottom layer of uncompacted waste also facilitates temperature increase.

(5) Sewage sludge and/or old anaerobically degraded refuse may improve methane generation from landfills. It is possible that the co-disposal of abattoir waste at the Grahamstown landfill has had beneficial effects.

(6) Controlled leachate recycle may enhance methane generation in areas with a low mean annual rainfall. Excessive recirculation may have negative effects. Leachate recycle at the Grahamstown landfill has not had any visible effects apart from improving containment of the LFG and removing the leachate problem.

(7) Efficient containment of LFG is a pre-requisite to the enhancement of methane generation. LFG at the Grahamstown landfill is inadequately contained during dry conditions because of cracking of the top surface.

5. A NEW APPROACH TO LANDFILL MANAGEMENT

Using data collected from numerous laboratory experiments and lysimeter tests, Stegmann and Spendlin[39] have developed a landfill operation strategy aimed at enhancing the landfill degradation process. It differs from the conventional approach

of compacting the refuse to as great a degree as possible (in order to prolong site life, prevent fires and rodents) and covering the refuse with soil or clay as it is compacted (to prevent wind-blown litter and flies). Initial results from the Lingen Landfill, Germany, indicate that this new approach may indeed be successful in accelerating waste decomposition[39].

The operation techniques and their consequences can be summarised as follows[39]:

(1) The first layer of refuse (1.5 - 2 m high) which is placed at the bottom of the landfill should not be compacted. This bottom layer undergoes rapid aerobic decay. Aeration may be induced by the placement of perforated pipes to facilitate the composting process. The advantages of this operation are that (i) the high organic leachate concentrations from subsequent upper layers of refuse can be degraded in this partly composted layer and (ii) the high temperatures developed in this layer can heat the layers of refuse above, ensuring that optimum temperatures (35-45°C) are attained. Disadvantages relate to (i) high leachate production rates during the first 6-12 months after placement, because of the high infiltration rates in the uncompacted refuse and (ii) the reduction in material available for methanogenesis, if commercial gas extraction is envisaged; although it is believed that acceleration of the decomposition process will ultimately increase the methane yield.

(2) After the organic concentration of the leachate has

decreased to a small percentage (after a period of 6 - 12 months), refuse should be placed in thin, highly compacted layers (30-40 cm) on top of this uncompacted layer. No cover material should be used if possible. If cover material is required to prevent wind-blown litter or flies, a very thin layer of permeable material should be used. The reason for placing refuse in thin layers and avoiding cover material, is to ensure efficient moisture infiltration and distribution within the refuse.

(3) Controlled leachate recirculation should be practised if precipitation is less than 800 mm per year. This will increase not only the moisture content of the refuse but also the substrate requirements of the methanogenic bacteria. The requirements for leachate treatment will be reduced as the leachate is effectively treated in the landfill itself.

The advantage of this type of management for gas extraction lies in the fact that the decomposition process is now reasonably controlled. The prediction of the time and period of methane production can thus be fairly realistically estimated. Stegmann and Spendlin[39] are of the opinion that methane production will occur 1-2 years after placement and that the period of gas production will be significantly shorter compared to that in conventional methods of management. A concentrated and efficient production of methane is to be preferred over an unpredictable and drawn-out one, not only from a gas exploitation point of view but from the point of view of environmental gas monitoring.

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CHAPTER 5

DESIGN OF LANDFILL GAS EXTRACTION SYSTEMS

CONTENTS

1. INTRODUCTION

2. GAS WELL DESIGNS

    2.1 Vertical Wells

    2.2 Horizontal Wells

    2.3 Comparison of Well Performance

3. GAS EXTRACTION SYSTEMS

4. THE GRAHAMSTOWN LANDFILL GAS EXTRACTION PLANT

1. INTRODUCTION

The objective of any gas extraction system is to extract the maximum volume of landfill gas using the lowest possible suction in the most cost effective way. If the gas is being extracted for commercial ends, it is further required that the ingress of air into the refuse is prevented. This chapter covers the practical aspects of the installation of a LFG extraction system.

2. GAS WELL DESIGNS

The first step in the construction of a LFG recovery system is the drilling and installation of extraction wells. There are two principal types of extraction well: vertical extraction wells and horizontal gas extraction trenches. The best type of well to install is determined by site specific factors such as depth of the landfilled waste, leachate levels and the method of landfilling, as discussed in section 2.3.

2.1 Vertical Wells

The traditional method of extracting LFG is via a series of vertical wells inserted into the refuse. A typical well design is shown in Figure 1. Initially a borehole is drilled into the refuse to a depth of 50% to 100% of the refuse thickness, the relative depth being determined by the depth of the landfill. Leachate levels dictate depth of extraction wells, Stegmann[9] being of the opinion that it is unnecessary to drill wells to the

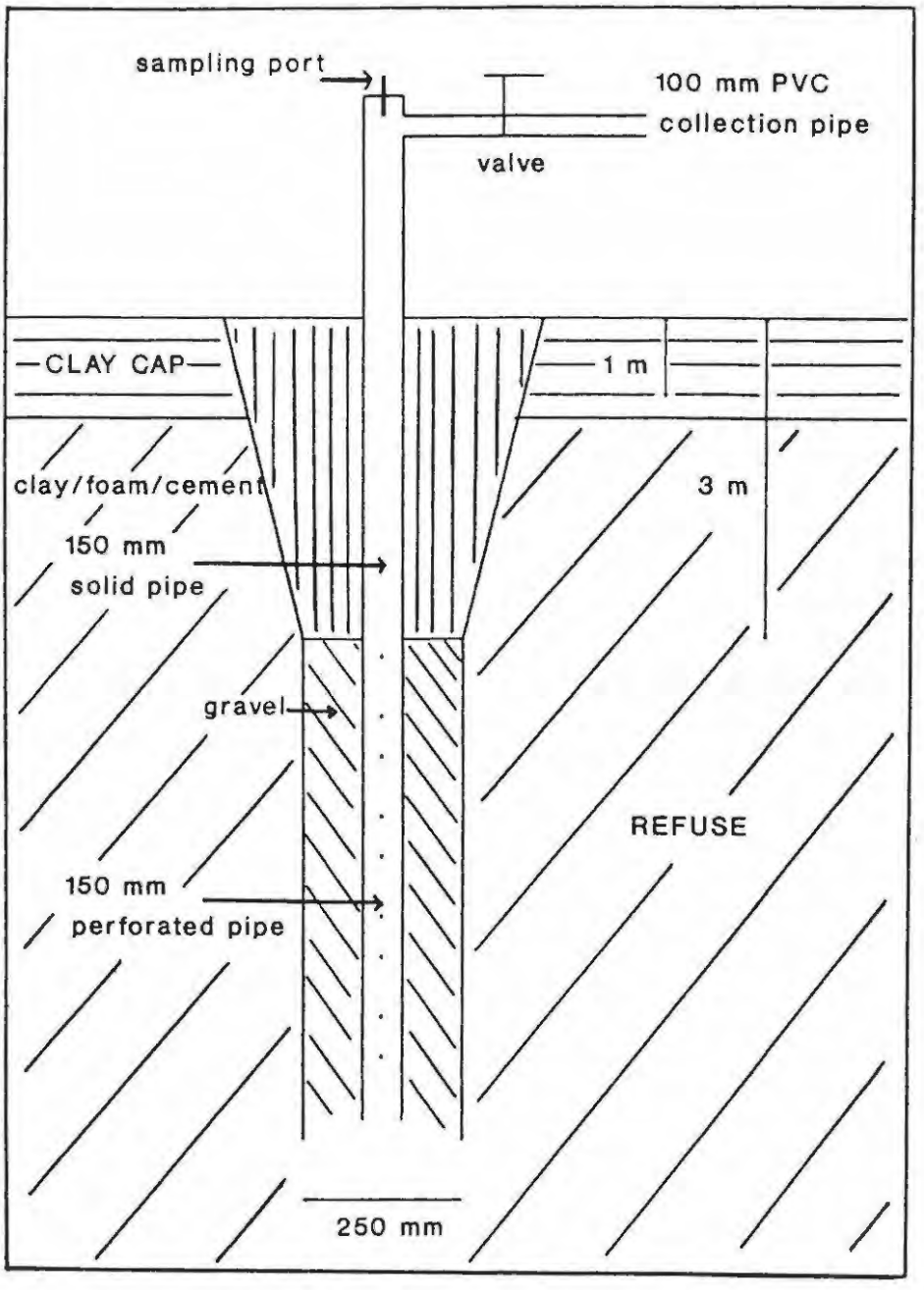

Figure 1. Vertical Gas Extraction Well (Adapted from Leach, 1991[7]).

base of a site as the bottom sections of the well are likely to fill up with leachate.

Well borehole diameters typically range from 100 to 900 mm, with the most common bore being in the range 300 to 600 mm[4,7,9]. It is generally considered that the larger diameter of the borehole, the more productive the gas well is[7]. This is related to the fact that larger diameter boreholes provide a greater surface area of refuse at the interface between the borehole and the refuse, and therefore require a smaller applied suction to attain a given LFG flow rate than do smaller diameter boreholes[4].

A gas extraction pipe, normally polyvinyl chloride (PVC) or high density polyethylene (HDPE), perforated except for the top 3-5 m, is inserted into the drilled borehole[1]. The higher the expected LFG flow rate, the greater should be the diameter of the piping[4]. Experiments at the Brogborough Landfill have shown pipe diameters of greater than 100 mm to give the best performance. In piping of smaller diameters, gas flow is restricted and higher suctions are required[7]. The Environmental Protection Agency (USA)[5] suggests that the pipe be perforated by 10 mm holes, 150 mm apart. Perforations in the form of sawed slots are also used[4,7,8].

The borehole is filled with gravel/chipped stone around the pipe, apart from the top 1-4 m. The top few meters are then filled with a plug of bentonite clay, cement, polyurethane foam or a combination of these, in order to provide a good seal against air

ingress[1,7]. Although cement is commonly utilised as a plugging material, its use is not recommended by the author. The cement may crack with subsidence of the refuse (as witnessed at the Grahamstown Landfill) thus permitting the ingress of air. A flexible material such as bentonite clay should rather be used.

Each well is connected to the collection pipework, preferably with flexible piping to allow for differential settlement rates of the refuse[7,9]. The well head should have a suitable valve to allow for adjustments to the suction on the well and to allow for isolation of the well. It should also have a sampling port to enable gas composition and flow rate to be measured (Figure 1).

2.2 Horizontal Wells

These are normally constructed by excavating trenches which are 1 m deep and 1 m wide across the refuse surface[6]. The trenches should have a slope of 2-5 in 100 to avoid pipes and ditches getting filled with water/leachate/condensate[11]. The trench is filled with gravel to a depth of 150 mm and a perforated pipe 50-150 mm in diameter placed on top. The pipe is covered with gravel/rubble and the trench backfilled with refuse. Unperforated pipes connect these trenches to a central well head, which should have the same facilities as those discussed for the vertical well[11]. A horizontal well is schematically depicted in Figure 2.

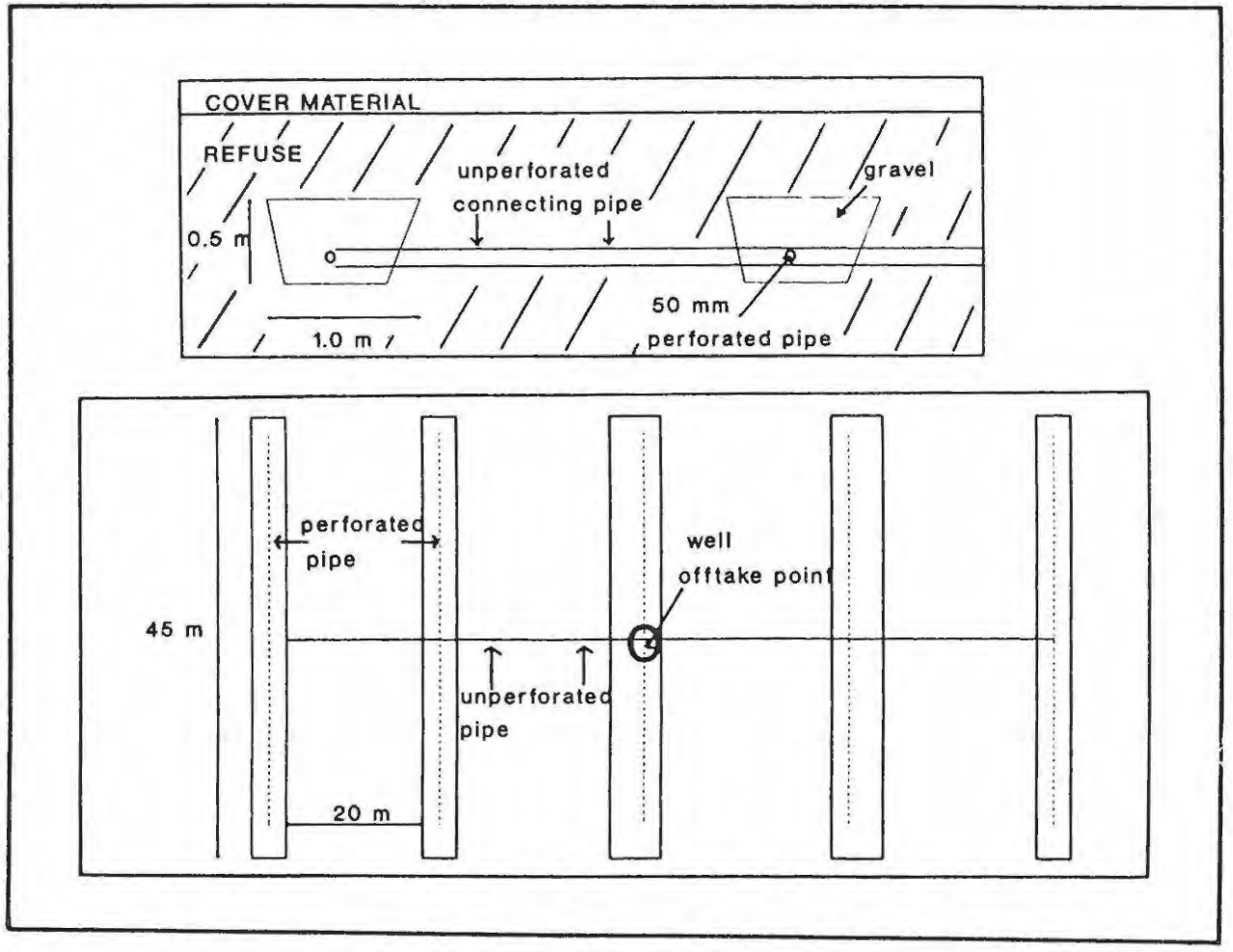

Figure 2. Horizontal Gas Extraction Trench (Adapted from Leach, 1991[7]).

2.3 Comparison of Well Performance

Both well designs pose different problems as regards effectiveness, cost and operational disturbance.

Initial decisions on which of the two types of well to install should be based on whether or not it is desired to extract LFG from a site which is still operational or from one where the waste is at a final level. Vertical wells are generally installed in sites which have been brought up to final level. Unfortunately, this entails a loss of a significant quantity of LFG prior to installation. The practical difficulties experienced in progressively extending vertical wells as the level of waste rises does make the horizontal trench method very attractive[6,7]. There are, however, documented cases of the use of vertical wells in operational sites. The Enderby Warren Landfill in Leicester (UK) is a case in point and has 39 extraction wells installed in the operational area. The connecting piping is flexible and is supported above ground by scaffolding. Wells are extended as the waste level rises with solid piping[10]. Four wells in the Grahamstown Landfill Site are also installed in the operational area, as discussed in section 4 of this chapter.

Horizontal wells can easily be installed while tipping proceeds, allowing for the extraction of LFG from lower waste levels while waste is being deposited at higher levels. Their installation does, however, depend on the method of landfilling. If a site is filled in large areas, so that its depth is built up slowly,

horizontal trenches can easily be installed, providing careful machine operators are employed. If, however, the current method of landfilling in small cellular areas is followed, the installation of horizontal trenches is not really practical, because many short lengths of piping are required[6]. It is impossible to install horizontal trenches once the site has reached final level, because of the practical problems of digging the trench, the ingress of oxygen into the refuse, the considerable odour nuisance and the loss of waste density[2,7]. Trenching through a final cap is not advisable as this creates a permanently weak link in the cap[2].

The effect of high leachate levels or refuse water tables* on the performance of vertical wells is extensively documented[2,6,7,8,9,11]. The higher the leachate level, the poorer the performance of the well, due to the reduction of the length of perforated piping through which the gas can be drawn, as gas cannot be sucked through saturated waste[7]. This phenomenon has been encountered at the Grahamstown Landfill: in two of the extraction wells no LFG was detected when leachate rose to above the level of the perforated piping (see Chapter 6). Blocking of wells by leachate is more likely to occur in shallow sites, where wells have short sections of perforations[6], and in wet areas with perched water tables. The solution to this problem is to either (i) re-drill the wells and install piping which is perforated to a higher level or (ii) remove the leachate from the

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*The terms "leachate level" and "refuse water table level" are synonymous terms. The former term will be used throughout this report.

gas well. Willumsen[11] has designed a well which contains a drain pipe within the gas suction pipe for this purpose. The methods commonly available for leachate removal from extraction wells have been evaluated by Leach[7]. The performance of horizontal wells is often better than that of vertical wells under conditions of high leachate levels. The reason for this is that in spite of the trenches at lower levels being blocked by leachate, extraction can simply be confined to trenches in the upper, unsaturated layers of waste[6].

While specialist drilling contractors are required for installing vertical wells, horizontal trenches can be installed with semi-skilled labour. However, for a deep site, the overall cost of installing horizontal trenches may be excessive in terms of the length of piping required.

Vertical wells tend to "rise" above the final waste level as settlement of the waste occurs. If something prevents the well from rising, a cement plug for example, it will buckle, usually in its mid-section[7]. The connections from well heads to the horizontal gas collection system may also break, if differential settlement rates between the horizontal and the vertical are experienced. For this reason, flexible piping must be used for the connections, and a flexible clay plug for the well seal[7,9]. Vertical pipes have also been reported to be compressed as a result of settlement[11]. Horizontal gas wells are also readily broken by differences in settling rates unless flexible coupling is used[9].

Leach[7] has determined that vertical wells, in the absence of high leachate levels, perform better than horizontal wells. This has been explained by the fact that the lateral permeability of refuse is as much as 40 times greater than the vertical permeability, due to the practice of putting waste down in layers and covering it with soil/clay. Vertical wells penetrate all layers and therefore take advantage of higher lateral permeabilities. (The results of measurements conducted to ascertain to what extent LFG is contained in the Grahamstown Landfill, Chapter 4, section 3.2.4, indicate that lateral permeability is indeed higher than vertical permeability. Methane concentrations of 5% were detected in the sides of the site, while a maximum of 4% methane was detected in the top surface.)

It should be evident that the decision as to which well type to install at a particular site depends on a number of site specific parameters. For example, a deep site which is at final restoration level will require vertical wells, while a shallow site covering a large area and still in its operational phase might require horizontal trenches[6].

3. GAS EXTRACTION SYSTEMS

Once the decision as to what type of well to install has been made, an assessment of the number of wells required to extract the estimated quantity of LFG produced by the landfilled refuse and the spacing of these wells is the next step. The assessment of LFG potential and the corresponding number of vertical wells

required to extract this LFG (related to the "cylinder of influence " of each well) is discussed in Chapter 10. Horizontal well spacing can also be calculated from determination of the "cylinder of influence"[5].

As a general guide, vertical wells in the interior of the site should be spaced 20-30 m apart and LFG extracted at a negative pressure of 1-3 cm of water, the suction being determined by the natural flow rate of LFG from the wells. The applied negative pressure should be only slightly greater than the positive LFG pressure inside the extraction wells. Wells at the perimeter of the site should be spaced closer together than wells in the centre of the site, and very low suction levels employed in order to prevent the possibility of air ingress through the sides of the site.

Each of the extraction wells in a site should be joined together and connected to a vacuum pump/blower/compressor via unperforated piping that may either lie on the surface or be buried in shallow trenches below the surface. The piping should be sloped by at least 2 in 100 in order to facilitate the drainage of condensate to specially designed condensate drains[9,11]** (see Figure 3). Before gas enters the pump/blower/compressor it should be cooled, in order to remove moisture. After leaving the pump it should

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**LFG is usually saturated with water vapour. Because temperatures at or near the surface of a landfill (10-25°C in SA) are lower than those deeper down in the waste (25-35°C in the Grahamstown site), the water vapour in the gas condenses when the gas is extracted. This condensate can block gas collection pipes unless the system is properly engineered.

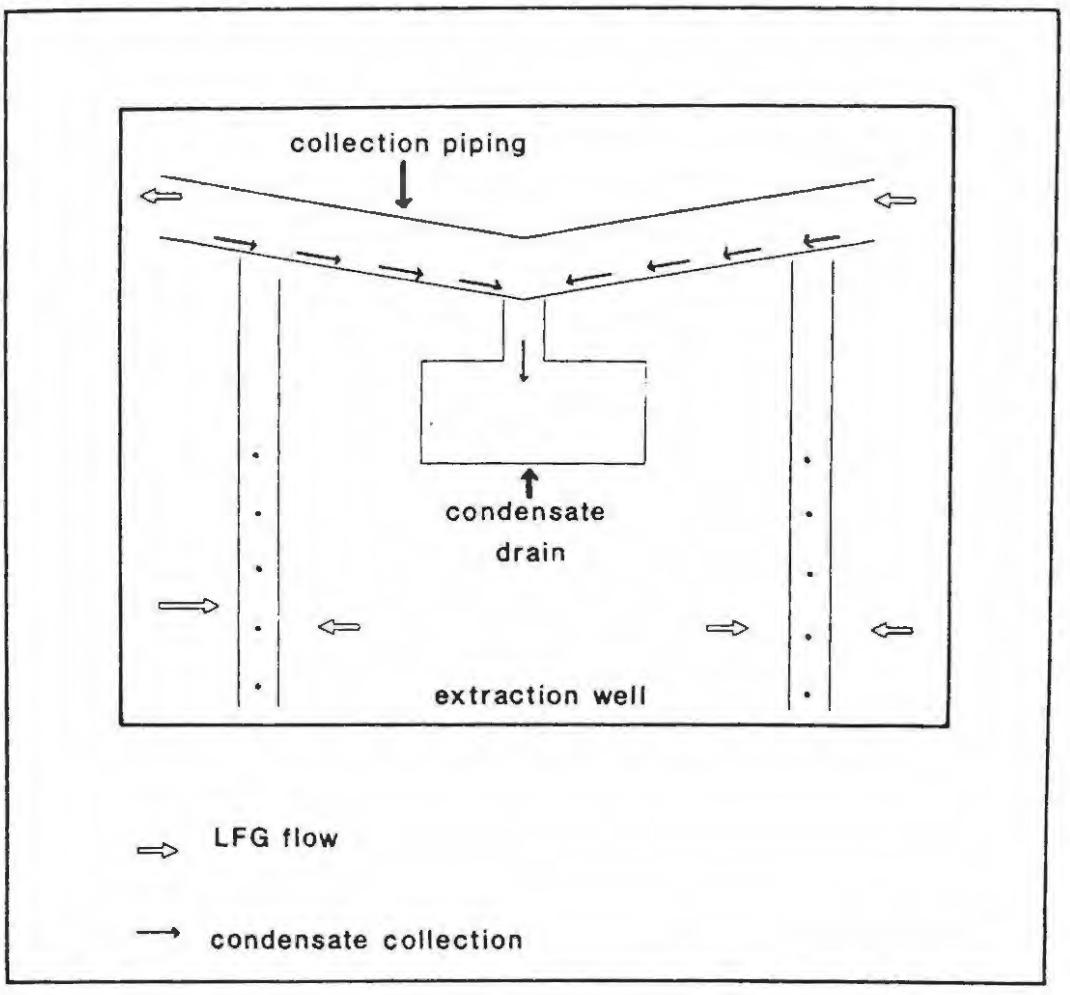


Figure 3. Condensate Drain (Adapted from Willumsen, 1991[11]).

pass through a flame trap before being fed to any utilisation options, so as to prevent the possibility of flash backs[3].

4. THE GRAHAMSTOWN LANDFILL GAS EXTRACTION PLANT

The siting, history and operation of the Grahamstown Landfill has been discussed in Chapter 4, section 3.

The initial LFG extraction system, consisting of six vertical wells, was installed in August 1988. Four of these wells were sunk in the rehabilitated portion of the site, wells A1, A2, A3 and A4 and two in the present tipping site, wells B1 and B2. The layout of the well field is schematically depicted in Figure 4.

During August 1990, the wells were re-drilled (250 mm bore) with a simple percussion rig, normally used for drilling water boreholes, to a depth of 10-12 m. Figure 5 and Photograph 1 are examples of such a well. The piping in the wells consists of 75 mm diameter flexible perforated piping, conventionally used as drainage piping (Photograph 2), for the bottom 5 m and solid "class 9" PVC for the top 6 m. The PVC piping is perforated with 10 mm diameter drilled holes up to 3 m from the surface. The extraction wells were backfilled with gravel up to 1 m from the surface and the well sealed with cement. (It has subsequently been realised that cement is not a suitable material to use). The well heads consist of a sampling port and a valve in the 75 mm PVC collection pipe joining the wells to the pumphouse.

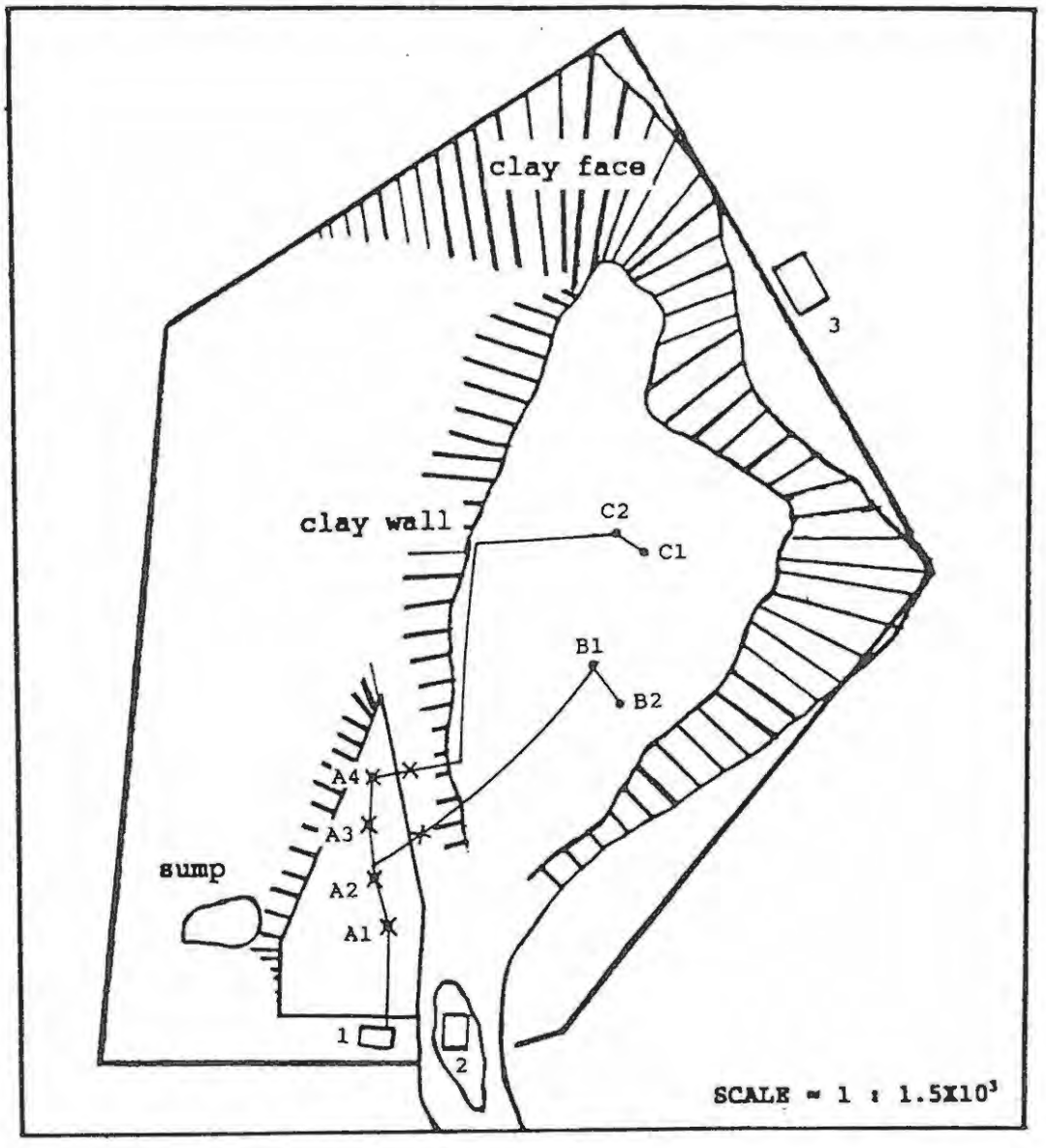


Figure 4. Layout of Grahamstown Landfill Well-Field

1 - pumphouse
2 - office
3 - caretakers' cottage
X - control valves
— - collection piping

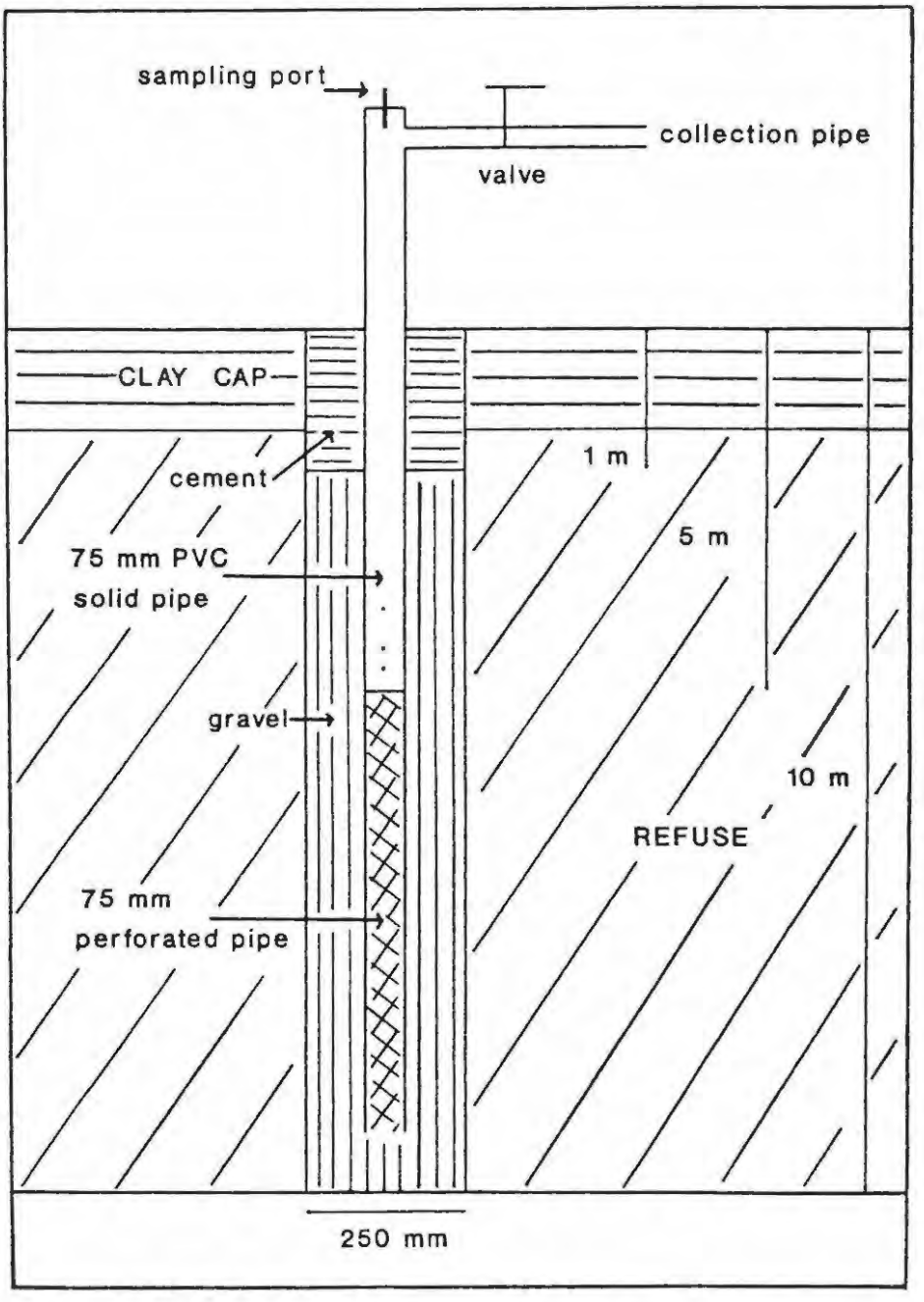


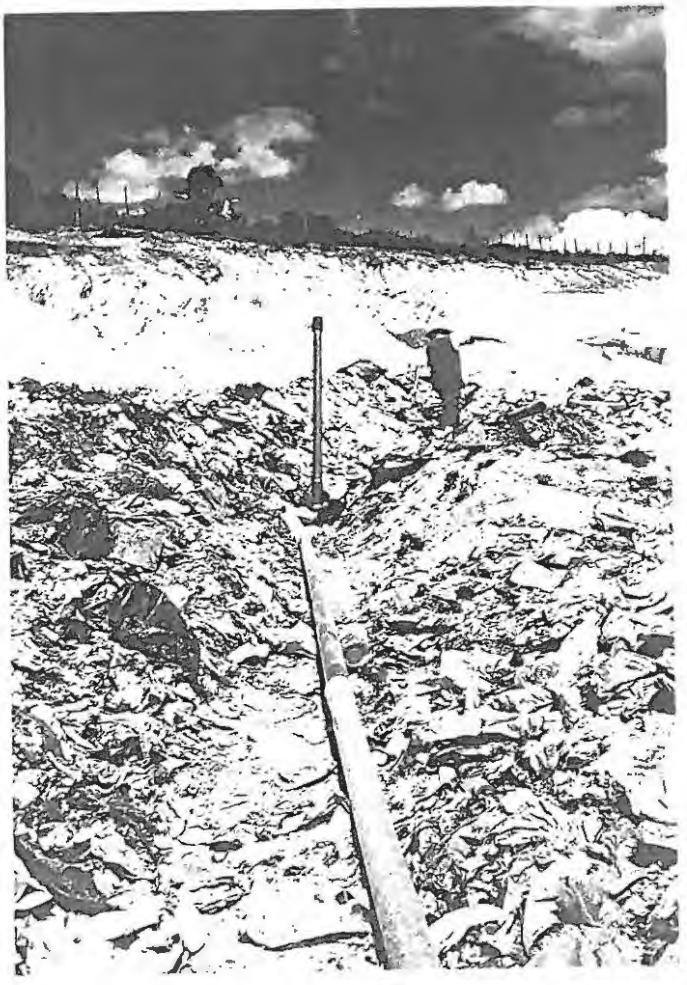
Figure 5. Gas Extraction Well at the Grahamstown Landfill Site.

An additional two holes, 15 m deep, have been sunk in the present tipping site, wells C1 and C2 (see Figure 4, Photograph 3). The wells are connected to the main extraction line with solid 75 mm PVC interspersed with flexible corrugated piping at intervals along the line. This was done to overcome problems related to pipe blockages and breakages from possible future subsidences.

As tipping continues, wells B1, B2, C1 and C2 are extended by simply placing a plastic bag over the well opening, as a seal against air infiltration and/or methane diffusion, and joining on an additional length of perforated piping. The refuse is then built up against the pipe in order to support it. This method is not ideal as no chipped stone is used around the pipe exterior. The perforations can thus easily become blocked by refuse. A better method would be to place a pipe of a larger diameter over the perforated pipe, fill up the space between the inner and the outer pipe with chipped stone, and then remove the solid outer pipe as the waste level rises.

The absence of condensate drains in the pipework has proved to be a limitation of the extraction system, as pipes do become blocked by condensate. Sections of the pipes have to be periodically excavated and the condensate removed.

The actual process of LFG extraction and subsequent utilisation is discussed at length in Chapter 11.



Photograph 3. Gas Extraction Well in the Present Tipping Face.

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CHAPTER 6

MONITORING OF LFG COMPOSITION

CONTENTS

1. OBJECTIVES

2. METHODS

3. RESULTS AND DISCUSSION

    3.1 Wells A1, A2, A3 and A4

    3.2 Wells B1 and B2

    3.3 Wells C1 and C2

4. CONCLUSIONS

1. OBJECTIVES

Monitoring of methane, carbon dioxide and oxygen concentrations in the LFG from the Grahamstown Landfill Site was routinely carried out for a period of a year (July 1990 - June 1991), with the purpose of (i) confirming the existence of an anaerobic phase of refuse decay, (ii) establishing the energy content of the LFG and (iii) detecting possible air ingress into the site and/or the pumping system. Only these three gases were monitored for; the concentrations of nitrogen and trace components, which make up the rest of LFG, (see Chapter 3) were not determined.

2. METHODS

Monitoring was carried out on all of the gas extraction wells; wells A1, A2, A3, A4, B1, B2, C1 and C2 (see Figure 4, Chapter 5); on a weekly basis. A battery driven, adapted motor car air-conditioning fan was attached to each well-head and gas extracted from the well at a pressure of approximately 2 cm water (12 $m^3.hr^{-1}$ LFG). This extracted gas was directly fed into portable gas detectors (Photograph 1).

Methane was detected with the use of two instruments: (i) a hand-held methanometer designed for the detection of gas in mines and (ii) a GMI oxygas-gascoseeker. Both these instruments are based on thermal conductivity detection (see Chapter 3, section 1.3.2). Oxygen was also monitored with the GMI oxygas-gascoseeker, which detects oxygen with the use of an oxygen electrochemical cell

Photograph 1. Monitoring of LFG Composition.

(see Chapter 3, section 1.3.7). Carbon dioxide was monitored with a GMI instrument based on infra-red detection (see Chapter 3, section 1.3.4). The LFG was extracted from the wells for at least 5 minutes before any measurements were taken.

The concentrations of components were detected as a percentage of the total gas volume (% by volume), and not as "absolute" concentrations (eg. $mg.l^{-1}$). Because the values are not absolute values, they cannot strictly be compared with one another (pers. comm. E. Senior). Only general trends can be noted*.

3. RESULTS AND DISCUSSION

3.1 Wells A1, A2, A3 and A4

Average monthly methane and carbon dioxide concentrations of the LFG from wells A1, A2, A3 and A4 are graphed in Figure 1. The readings taken during October 1990 are not included because during this time numerous exhaustive pumping experiments (Chapter 11) were carried out and levels therefore fluctuated considerably. Oxygen concentrations are not given because levels are consistently low, 0% for wells A2 and A3, and 1-3% for wells A1 and A4 (with some exceptions, which will be commented on). It can be assumed that the rest of the gas is largely made up of nitrogen, from air. On the basis that air contains 80% N_2 and 20%

*To illustrate this point, consider for example that under dry conditions, LFG is monitored to contain 60% CH_4 and 40% CO_2. Under wet conditions, the CO_2 can dissolve, and as a result might only be 20%. Methane would then be 80%, even though the absolute concentration of methane might not have changed.

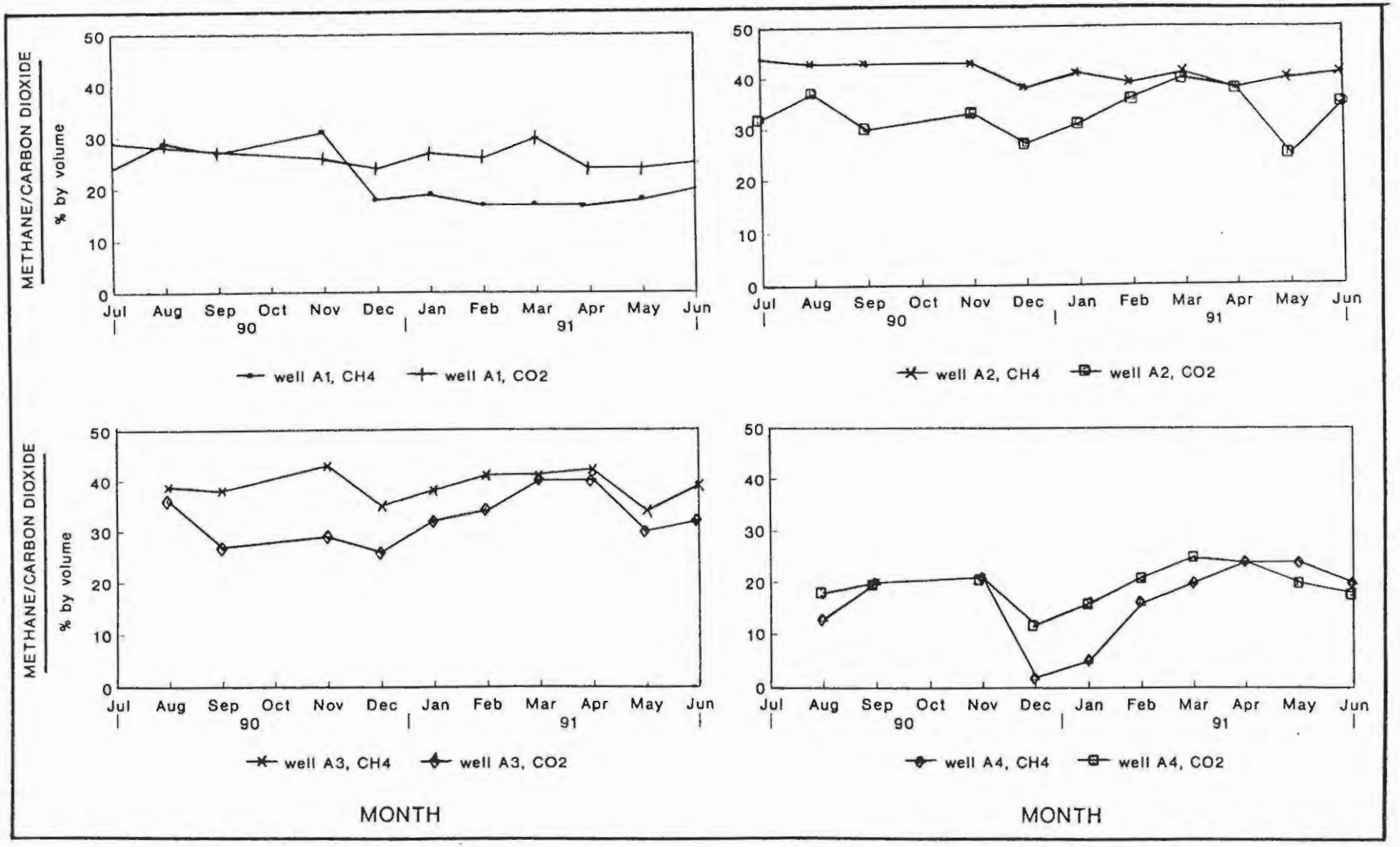

Figure 1. Methane and carbon dioxide concentrations in wells A1, A2, A3 and A4 for the period July 1990 - June 1991.

O_2, the oxygen concentrations measured are lower than expected. This is because oxygen is readily consumed by bacteria as it is drawn through the refuse.

For purposes of comparison, Figure 2 shows methane concentrations from all four wells and Figure 3 shows carbon dioxide concentrations from all four wells.

From Figures 2 and 3 it is observed that wells A2 and A3 are very similar, with relatively stable methane contents, 40% on average, and carbon dioxide contents averaging 35%. This is unlike wells A1 and A4 which fluctuate considerably in gas composition and have lower methane and carbon dioxide concentrations than wells A2 and A3 at all times. This is possibly related to the fact that wells A1 and A4 are located very close to the edges of the rehabilitated site ($\approx$ 2 m), whereas wells A2 and A3 are in more central positions. Air is easily pumped into the refuse in the areas of wells A1 and A4, inhibiting the methanogenic process. This is verified by the fact that the oxygen concentration in the LFG from wells A1 and A4 is, on average, between 1 and 3%, while that in wells A2 and A3 is normally zero.

It is noteworthy that for the months of December 1990 and January 1991, the methane concentration in well A4 approached zero (and oxygen reached 20%). This is related to an incident in which the plug which normally seals the well-head, was mistakenly not screwed into the well-head while gas was being extracted from the well-field with the main pump. Air was thus sucked into the site,

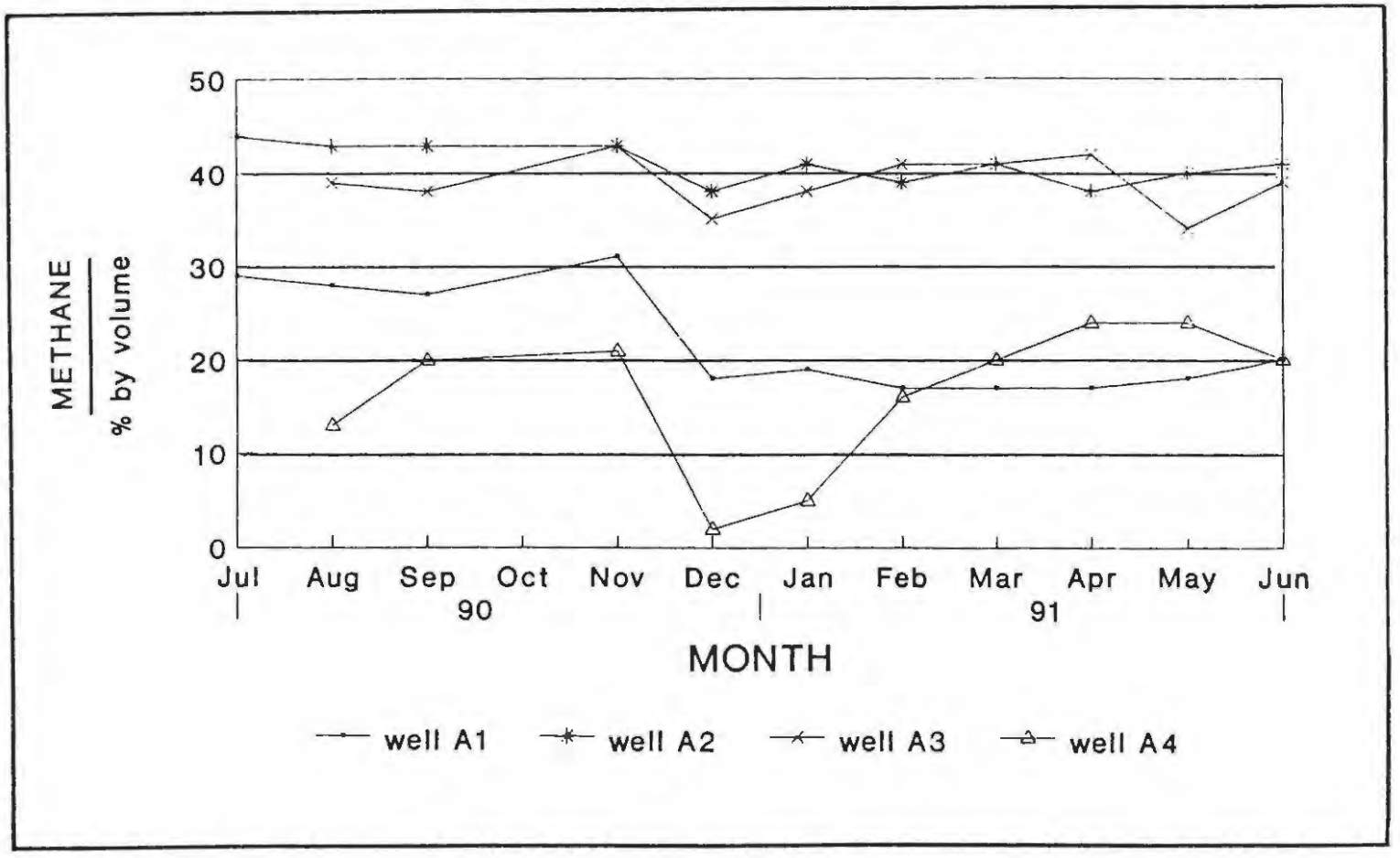

Figure 2. Methane concentrations in wells A1, A2, A3 and A4 for the period July 1990 - June 1991.

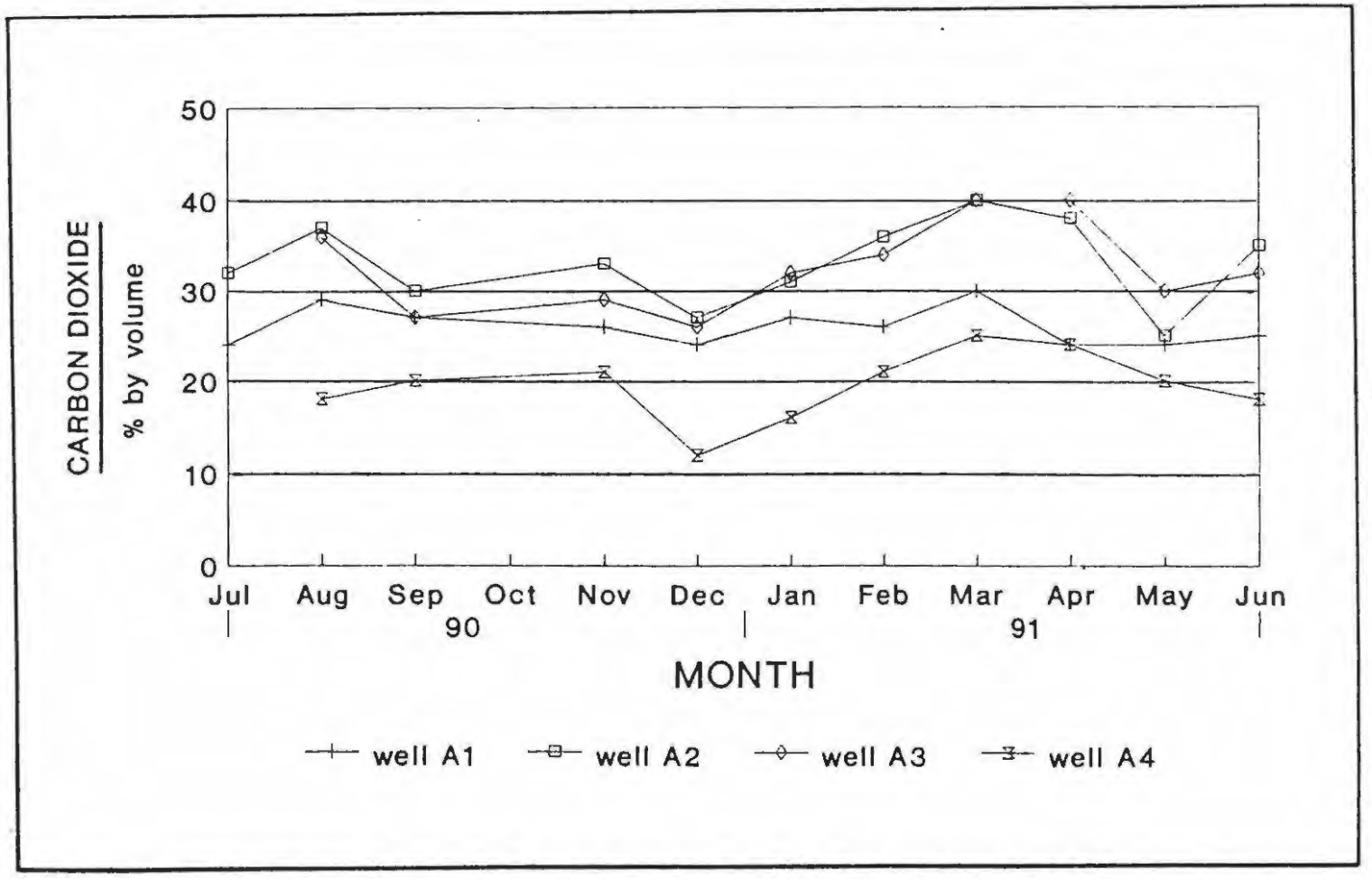

Figure 3. Carbon dioxide concentrations in wells A1, A2, A3 and A4 for the period July 1990 - June 1991.

poisoning the methanogens. The methane concentration in all the wells in fact dropped during December 1990. By February 1991, (2-3 months later), methanogenic conditions appear to have been re-established in well A4.

Methane concentrations are higher than the concentrations of carbon dioxide in the LFG from all wells, except in wells A1 and A4, where carbon dioxide content is sometimes greater than methane content. This is probably related to air ingress into the refuse in the vicinity of wells A1 and A4, which promotes aerobic rather than anaerobic decomposition.

Variation in methane concentration with time seems to follow a generally similar pattern for wells A1, A2, A3 and A4, indicating that there is possibly some external factor which is controlling biogas composition. In order to investigate this possibility, ambient temperatures (averages of the monthly maximum) and monthly rainfall were recorded[4] over the period of investigation and compared with variations in methane and carbon dioxide concentrations of the LFG from wells A2 and A3 (Figures 4 and 5).

While there does not appear to be any definite correlation between trends in either ambient temperature or rainfall, and the methane content of the LFG from wells A2 and A3, it is apparent that the lowest methane concentrations (December 1990 - May 1991) are recorded when ambient temperatures are highest. In order to investigate whether changes in ambient temperature have an effect on the refuse ecosystem, landfill temperatures were measured, at

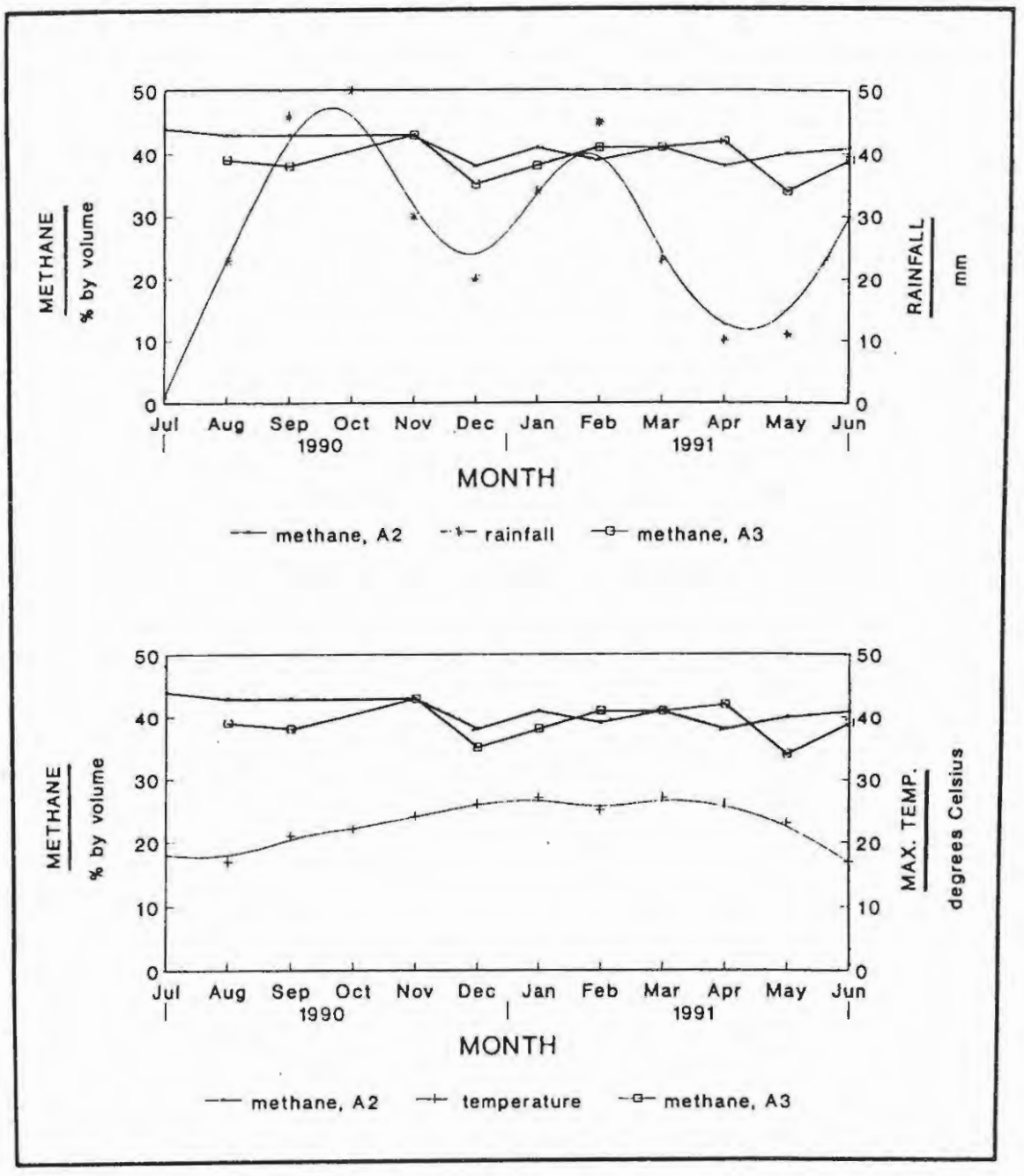

Figure 4. Methane concentrations in wells A2 and A3, and monthly rainfall and average maximum ambient temperatures for the period July 1990 - June 1991.

the LFG.

3.2 Wells B1 and B2

Prior to the drilling operations in August 1990, during which the wells were re-drilled and new piping put down (Chapter 5), the methane concentration in well B1 averaged 42% and the carbon dioxide concentration 25%. For two months after the installation of the new piping, the concentrations of both methane and carbon dioxide were zero, and the oxygen concentration was 21%. The well has not been monitored since October 1990 because it has no well-head. This is due to the fact that it is being continually extended as tipping operations proceed (see Chapter 5).

It is not known why bacterial activity should have appeared to cease (as indicated by the total absence of methane and carbon dioxide in the LFG) once well B1 had been re-installed. A very strong smell of PVC glue was noted in the gas pumped from this well once it had been re-installed. The possibility exists that bacteria were poisoned by toxic organic compounds in the glue used during pipe installation.

During July 1990, well B2 exhibited an average methane concentration of 20% and a carbon dioxide concentration of 12%. At the end of July, prior to drilling operations, the concentrations of both gases approached zero. It was discovered that the well was completely filled with leachate/water. (It is possible that this was related to the location of the well, which

is in close proximity to an old abattoir pit.) Another well was drilled 10 m away from the saturated well. Methane concentrations increased to 40% within a month, but dropped to zero within the next month, when this well also became flooded. The well has not been monitored since October 1991, for the same reasons that well B1 has not been monitored.

The indications are that either (i) methane is not produced from refuse that is below the water table or (ii) that gas extraction is not possible once high water tables fill up extraction wells. The first possibility, which is related to the biochemical phenomena of decreased pH, increased redox potential and cooling of the refuse ecosystem with increased saturation, has been discussed at length in Chapter 4, section 2.1. The second, which is related to the difficulty of physically drawing gas through saturated waste is covered in Chapter 5, section 2.3.

3.3 Wells C1 and C2

Both these wells were monitored from October 1990 (shortly after they'd been constructed) to January 1991 (when access to the wells was made difficult by the close proximity of the abattoir pit). Well C1 averaged a methane content of 42% by volume, and a carbon dioxide content of 30% by volume. Well C2, which is a mere 10 m away from well C1, had a 46% methane content and 35% carbon dioxide content. The LFG from the wells contained no oxygen.

The consistently high methane concentrations monitored are possibly related to the facts that (i) the wells are situated next to a pit which is used for the disposal of abattoir wastes and (ii) the refuse is young (< 3 years old). As regards the first fact, the abattoir waste provides a source of bacteria and nutrients, and can be regarded as a good "seed" for methanogenesis. This is similar to sewage sludge, which is well documented as a good seeding material[7,8,10]. Secondly, methane production is expected to peak 1-2 years after placement of refuse (depending on climatic conditions)[2]. Refuse in the vicinity of wells C1 and C2 is therefore probably in the most active stage of methanogenesis, as it is between 6 months and 3 years old.

4. CONCLUSIONS

The LFG composition data collected from gas extraction wells in the Grahamstown Landfill Site, reflects the multivariant nature of the landfill ecosystem. Composition appears to fluctuate at random and it can be concluded that a number of factors probably determine composition. There appears to be no unambiguous relationship between composition and either ambient temperature or rainfall. It has been established that (i) the proximity of gas extraction wells to the edges of the site and (ii) the level of the water table in the site, have some effect on gas composition. The age of refuse and the use of a seeding material such as abattoir waste may have an effect on composition.

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CHAPTER 7

MONITORING OF LFG

FLOW RATES

CONTENTS

1. INTRODUCTION

2. METHODS

3. RESULTS AND DISCUSSION

    3.1 Quantification of Flow Rates

    3.2 Relationship between Flow Rate and Atmospheric Pressure

4. CONCLUSIONS

1. INTRODUCTION

It is well documented that the pressure inside a landfill is usually higher than the pressure outside[1,2,3], as a result of the production of LFG. It follows that if the gas extraction wells in a landfill site which is well covered, are opened to the atmosphere, LFG will escape, at a rate determined by the rate at which it is produced in the site. This forms the basis of the monitoring of LFG flow rates at the Grahamstown Landfill.

Monitoring of the rate at which LFG escapes from the landfilled refuse at the Grahamstown Landfill Site, has been carried out in order to (i) determine a rate of LFG production and hence establish at what rate the gas can be extracted, (ii) ascertain whether or not there is any relationship between flow rate and atmospheric pressure and (iii) provide data for comparison with flow rate predictions obtained from modelling of the gas generation process (Chapter 10).

2. METHODS

Flow rate measurements have been carried out in two areas: on the gas extraction wells in the rehabilitated site; wells A1, A2, A3 and A4; and on the two wells in the present tipping area; wells C1 and C2 (see Figure 4, Chapter 5). Monitoring of the flow rate from each well was conducted while all the wells were open to the atmosphere. Measurements were taken at different times during the day and night, and the atmospheric pressure at the time of

measurement recorded using a conventional barometer.

The instrument used to monitor flow rates was a thermal anemometer, "TESTOVENT 4100", conventionally used in air conditioning monitoring. This instrument gives flow rate readings in units of $m.s^{-1}$. The units were converted to $m^3.hour^{-1}$ by measurement of the cross-sectional area of the piping through which the gas escaped, no account being taken of the non-uniform velocity distribution across the pipe.

3. RESULTS AND DISCUSSION

3.1 Quantification of Flow Rates

The flow rates determined from each gas extraction well in the two areas, have been summed to give values for the total gas flux from the two areas concerned. The values so obtained are detailed in Table 1.

The average rate at which gas escapes from the open wells in the rehabilitated site is very low, $\approx 2\ m^3.hr^{-1}$, reflecting a low rate of LFG production. More gas escapes from wells A2 and A3 ($\approx 1.2\ m^3.hr^{-1}$), than from the perimeter wells, wells A1 and A4 ($\approx 0.6\ m^3.hr^{-1}$). This is probably related to the fact that wells A2 and A3 exhibit a more developed state of methanogenesis than do wells A1 and A4, as indicated by the composition of LFG from the wells (see Chapter 6).

Gas escaping from wells C1 and C2, does so at approximately twice

the rate of that escaping from the wells in the rehabilitated site, ≈ 4 $m^3.hr^{-1}$. Methanogens are probably more active in this area of the landfill, a conclusion supported by the fact that the methane content of the gas (42%) is higher than that of gas from the rehabilitated site (35%). This may be related to the age of the refuse, 0.5 to 3 years in the first case, and 3 to 5 years in the second. The methane flux from the rehabilitated site is only 0.7 $m^3.hr^{-1}$, as opposed to a flux of 1.8 $m^3.hr^{-1}$ from wells C1 and C2.

TABLE 1
Flow Rates at the Grahamstown Landfill

WELL	LFG FLOW RATE $m^3.hr^{-1}$	CH_4 CONTENT % by volume†	CH_4 FLOW RATE $m^3.hr^{-1}$
A1+A2+A3+A4	2	35	0.7
C1+C2	4	42	1.6

NOTE: † See Chapter 6 for gas compositions of individual wells.

3.2 Relationships between Flow Rates and Atmospheric Pressure

Numerous studies[1,2] on the pressure of LFG in the cover materials overlying refuse in a landfill site have been conducted. Results indicate that changes in LFG pressures are related to changes in atmospheric pressure. During diurnal high pressures, air is drawn into the surface of the landfill cover material from the atmosphere, and LFG pressure in the cover material is negligible. During diurnal lows in atmospheric pressure, LFG moves upward to the atmosphere from the refuse, into the cover material, resulting in increased pressures in the cover material. The

overall effect is one of a continual flux of LFG in and out of the cover material[1,2,3].

It can be expected that if gas/air is allowed to move freely between the landfill/atmosphere interface, the relationship between gas pressure in the cover material and atmospheric pressure, will also hold true for LFG flow rates, i.e. the higher the atmospheric pressure, the lower the flow rates of LFG into the atmosphere and vice versa.

In order to investigate whether or not such a relationship exists, the results of flow rate measurements have been plotted as a function of atmospheric pressure (Figure 1). No correlation between gas flux and atmospheric pressure is evident. It could be that readings were taken over too limited a range of atmospheric pressures for any trends to be evident and/or that the barometric pressure measurements were not sufficiently sensitive.

The flow rate measurements from the rehabilitated site were similarly plotted versus the time of day (Figure 2) in order to detect any relationships of a diurnal nature. It is observed that flow rates peak during the hottest time of the day and are at their lowest during the evening and early morning. This is in agreement with the observation made by EMCON Associates[3] that internal landfill pressures fluctuate diurnally, with the highest and most stable pressures being reached in afternoon hours.

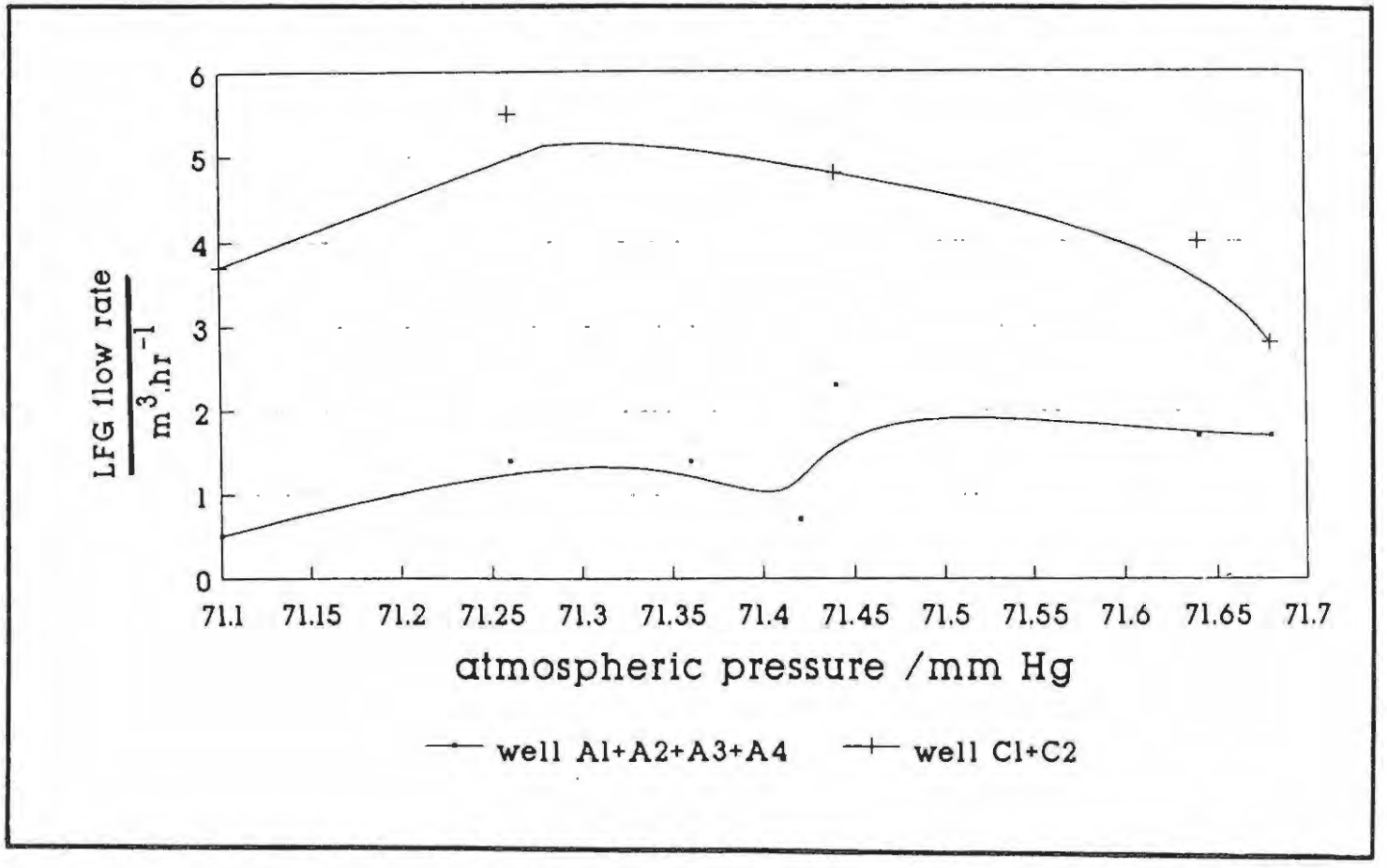

Figure 1. LFG Flow Rate versus Atmospheric Pressure

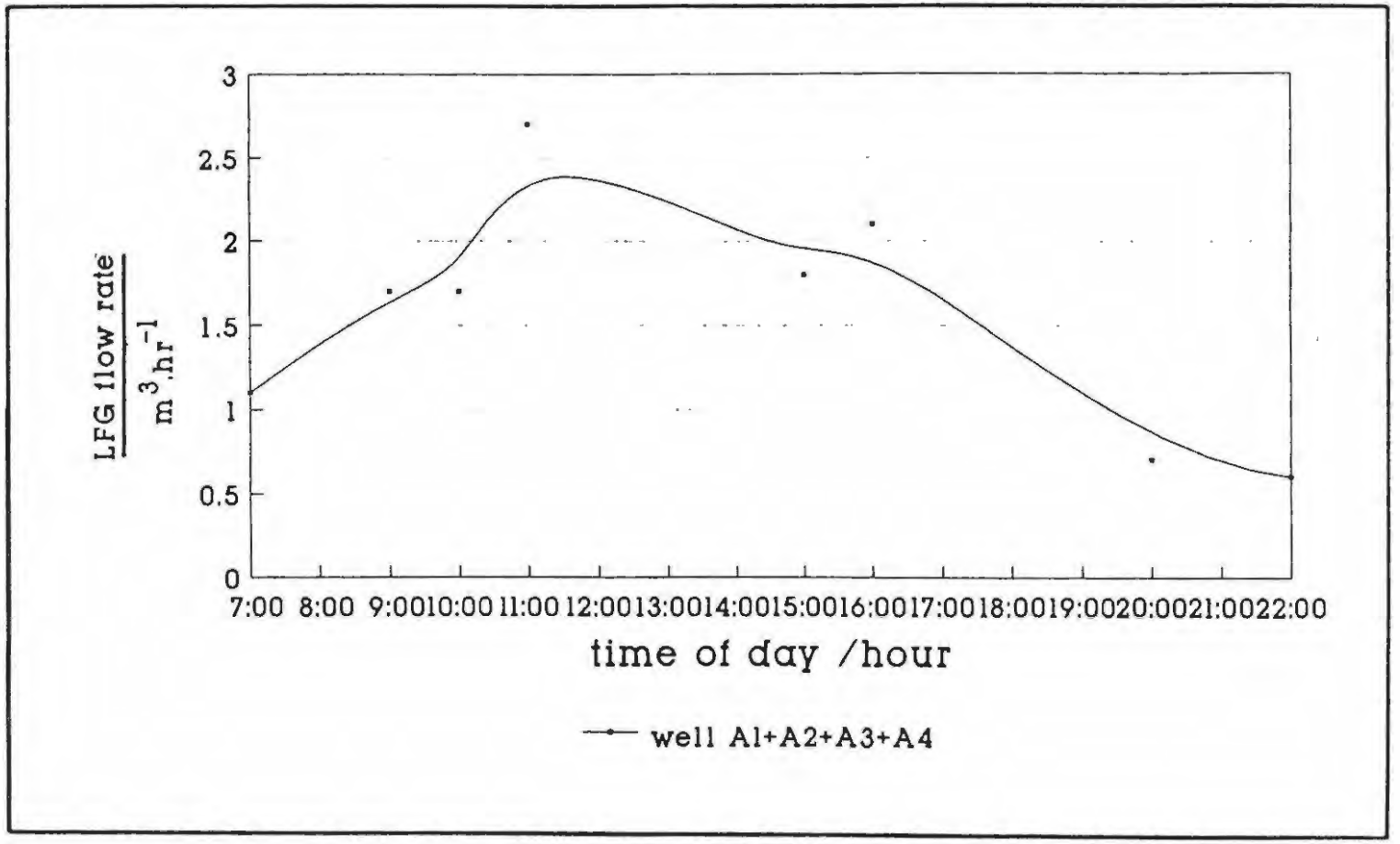

Figure 2. Flow Rate Fluctuations with Time of Day

It is a well established fact that atmospheric pressure and air temperature are inversely related. It can therefore be expected that LFG flow rates are lowest in the early morning and evening (low temperature and high atmospheric pressure) and highest in the afternoon (higher temperature and lower atmospheric pressure). It is rather paradoxical that while a relationship between flow rates and the time of day has been established for the Grahamstown Landfill Site, one for atmospheric pressures does not appear to exist. The possibility of some factor other than pressure controlling the observed diurnal relationship cannot be discounted.

4. CONCLUSIONS

The data reported in this study is limited by the methods used to obtain it. Gas flow rates were only measured from the open gas extraction wells and these readings summed to give the total gas flux from the rehabilitated site. It is very likely that these readings are in fact lower than the actual gas flow rates, as it has been established that some gas does in fact escape through the landfill cover and sides (Chapter 4, section 3.2.4). This contribution to flow rates has not been quantified, although it is believed to be negligible.

It is apparent that more measurements need to be taken, over a wider range of atmospheric pressures, before any conclusions concerning the relationship of flow rate to atmospheric pressure can be made.

It can tentatively be concluded that:

- LFG generation rates from the rehabilitated site, which has a volume of 6 500 m^3 and contains 5 000 tonne of refuse, are low ($\approx$ 2 $m^3.hr^{-1}$). This corresponds to a methane generation rate of approximately 0.7 $m^3.hr^{-1}$ or 1 $m^3.tonne^{-1}.yr^{-1}$ if the LFG detected is considered to be generated from all 5 000 tonnes of the refuse. The measured flow rates are compared with results of kinetic modelling of LFG generation in this part of the site in Chapter 10.
- LFG generation rates from wells C1 and C2 ($\approx$ 4 $m^3.hr^{-1}$) are considerably higher than for the rehabilitated site. If the LFG detected from the two wells is estimated to be generated from a refuse volume of approximately 3 000 m^3, the methane generation rate amounts to 7 $m^3.tonne^{-1}.yr^{-1}$. Projecting this onto the whole of the operational site (areas B, C and D, see Figure 3, Chapter 4), which has a volume of 77 800 m^3, the estimated LFG flow rate from this area is 104 $m^3.hr^{-1}$. If methane content is 42%, then the methane flow rate detected by this method is 44 $m^3.hr^{-1}$. The implications of this estimate with reference to the results of kinetic modelling are discussed in Chapter 10.
- LFG flow rates from refuse on the perimeters of the site is negligible.
- Gas flow rates exhibit a diurnal variation.

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CHAPTER 8

LANDFILL SITE INVENTORY:

REFUSE SURVEY

CONTENTS

1. PURPOSE OF SURVEY

2. KEY QUESTIONS

3. METHODS

4. RESULTS

5. COMMENTS

6. DISCUSSION

    6.1 Amount of Refuse in the Grahamstown Landfill

        6.1.1 This Survey

        6.1.2 Other Estimates

    6.2 Potential for Recycling

    6.3 Quantity of Biodegradable MSW

    6.4 Quantity of Biodegradable Domestic MSW

7. SUMMARY

NOTE: This chapter is written in the form of a report in which all the figures and tabulated data appear at the end of the discussion.

1. PURPOSE OF SURVEY

This survey was carried out in order to :

- Establish a data base for solid waste disposal practices in Grahamstown.
- Determine what fraction of the Grahamstown landfill is biodegradable. This will facilitate modelling of the decomposition process and allow predictions to be made as to :
  - the total amount of biogas potentially available;
  - the rate of gas production with time;
  - the time for the landfill to reach a "stable" state of equilibrium.
- Establish quantities of materials that can be recycled.

2. KEY QUESTIONS

The following questions form the basis of the survey :

(i) How much MSW (municipal solid waste; composed of domestic, garden, builders rubble, metal, paper, glass etc.) is disposed of at the Grahamstown Municipal Landfill Site?

(ii) What is the composition of this MSW?

(iii) What is the composition of the domestic refuse that is collected?

(iv) Of the MSW collected, how much :

(a) is landfilled

(b) is recycled

(c) is potentially recyclable

(d) is incinerated?

3. METHODS

The survey was conducted during the periods 22-26 October 1990 (Spring), 3-19 December 1990 (Summer), 20-26 March 1991 (Autumn) and 12-18 July 1991 (Winter). For the composition studies of domestic refuse, twenty filled refuse bags were collected at random each day from the area in which the municipality was collecting refuse on that particular day. These bags of uncompacted refuse were tipped into a 1 metre cubic box at the landfill site. The refuse was handsorted into 7 categories (paper, plastic, food, glass, metal, fines - a mixture of food waste and some paper, and miscellaneous - a fraction including textiles, rubber and polystyrene), and the various fractions weighed, on the same day. Composition was hence determined on a wet weight (ww)* basis.

Monitoring of the type and quantity of all the refuse entering the site was conducted during the same periods. Refuse was classified according to one of 8 categories (domestic, garden, builders rubble, metal, paper, glass, abattoir effluent and light

*Refuse composition on a wet weight basis is for refuse in the wet-as-received state and is abbreviated (ww). Composition on a dry weight basis can be calculated from the inherent moisture contents of refuse components and is abbreviated (dw).

industrial). Cover material obtained from the site itself was not included in the analysis. The quantity of refuse on the vehicles entering the site was usually judged by observing the tare of the vehicle and the extent to which the suspension was burdened. This approach was not required when monitoring the amount of refuse collected by either of the two municipal refuse collectors/compacters, as it is known that the smaller of the two vehicles carries a full load of 3.5 tonne while the larger carries a full load of 5 tonne.

4. RESULTS

Results obtained for domestic refuse composition on a wet weight (ww) basis, for each of the four seasons, are detailed in Tables 1, 2, 3, 4 and 5. Table 7 gives results on a dry weight (dw) (see Footnote*) basis. These were calculated using the assumed moisture content of components of MSW[6] given in Table 6.

Results of the monitoring conducted in order to determine the quantity and general composition of waste entering the site during each season, are detailed in Tables 8, 9, 10 and 11. The data is assumed to have an estimated error of 20%. This is attributed to the monitoring technique used, in which (i) the mass was simply estimated and (ii) observations were only made as to the general type of refuse that vehicles were transporting. No detailed analysis was carried out and the domestic fraction, for example, was not divided into components such as metal, paper and glass.

5. COMMENTS

There is general consensus among researchers as to the difficulty of ensuring the statistical significance of the data obtained in refuse composition surveys. In this survey, refuse bags were collected from 5 different areas, on different days of the week. Refuse composition over these 5 days varied considerably. On Friday the 26 October 1990, for example, when refuse was collected from a middle-class residential area, the paper fraction exceeded 40% and the organic fraction was less than 20%, by weight, of the total sample. On Monday the 22 October 1990, when refuse was collected from the CBD, the paper fraction was 40% and the organic fraction exceeded 20% (Table 1). This was most surprising, and demonstrates that preconceived ideas are not necessarily correct. The indication is that the survey should be carried out over a longer period of time.

The process of handsorting is far from an exact science. It became apparent in this survey that the significance of the data obtained was limited by the sorting method. It often became difficult, for example, to separate refuse components such as food contaminated newspaper and plastic.

The ideal method for determining the quantity of waste entering the site is by use of a weigh bridge. The small size of the landfill does not, however, warrant the installation of a weigh bridge.

The composition of the waste entering the site varied significantly from day to day during the survey periods (Tables 8, 9, 10 and 11). For example, the large quantity of soil (90 tonne) from building operations and excavations, deposited at the site on Wednesday the 24 October 1990 was quite exceptional. Refuse composition over the four seasons has been calculated (Table 12). This data has been further analysed and the domestic fraction divided into its various components (Table 13) using the data obtained from the domestic refuse composition studies.

Refuse quantity and composition varies with the time of year (Table 13). During this survey, more refuse was deposited during Spring than any other season, with the least being deposited during Summer; large amounts of builders rubble and soil, and garden waste were landfilled during Spring. This is not altogether surprising as the Summer survey was conducted during a holiday period when many residents and students vacate Grahamstown. The fractions of certain waste components did increase during Summer, notably the percentages of paper and metal. The increase in the percentage of metal is understandable given the large scale cleaning-up operations carried out by engineering firms during this time, but the increase in the paper fraction is somewhat of an anomaly if one considers that academic activities are significantly reduced during this time. Perhaps a lot of "tidying up" is done in offices at the end of a year. During Autumn and Winter deposited paper decreased by half that deposited in Spring and Summer. It is not known why this should

be so, although it is known that Rhodes University began actively to pursue a paper recycling programme at this time. The largest quantity of garden refuse was deposited during Autumn and is presumably related to pruning and the cleaning up of leaves.

The composition of MSW entering the Grahamstown Landfill is comparable to that of some of the studies conducted elsewhere (Table 14). The comparison of studies of this nature is complicated by the fact that (i) the procedures followed are not necessarily the same (ii) the categorisation of components is usually different and (iii) the populations involved vary in their standards of living and in the types of activities pursued. As such, refuse surveys are extremely site-specific.

6. DISCUSSION

6.1 Amount of refuse in the Grahamstown Landfill

6.1.1. This survey

On average, 68 tonne of wet-as-received refuse is disposed of every working day at the Grahamstown landfill (Table 13). This is equivalent to 18 000 tonne of refuse a year (assuming that there are 260 working days in the year). Given that the site is 5 years old, and assuming that the same amount of refuse has been deposited every year and that no refuse has been removed from the site, the site at present holds 88 000 tonne of wet-as-received refuse. If the inherent moisture contents of the components are

considered, this amounts to 61 000 tonne dry material.

6.1.2. Other estimates

In addition to the refuse survey, the amount of refuse in the Grahamstown landfill was estimated on the basis of (i) site dimensions and (ii) population estimates.

(i) The site dimensions were obtained from measurements taken on an aerial photograph (June 1991) and the photograph used to produce a scaled diagram of the site (Figure 1).The site volume was calculated (Table 20) and assuming an average compaction of 0.7 tonne.m^{-3}, the estimated quantity of in-place refuse was found to be 62 000 tonne. This estimate has not taken into account the fact that some of the site volume is taken up by the clay that is used as cover material, which is obtained from the site itself. The quantities of clay used are however minimal, with most of the cover material being provided by the soil which is deposited at the site as builders rubble (pers.comm. T.Pike).

(ii) The white population in Grahamstown, including students, amounts to 15 500 (pers.comm. R.Theron). If average refuse production, per person, per year, is taken as 0.365 tonne[3], the white residents produce some 5 660 tonne of refuse per year. The black population in Grahamstown is estimated at 60 000[7]. It is difficult to predict how much refuse each person in this community produces. It is most certainly

less than that for the white population. Due to this difficulty, an estimate as to the quantity of refuse produced by the black population, was made based on the number of vehicles depositing Rini (the non-white residential area) refuse at the landfill site. This is found to amount to 4 200 tonne.year^{-1}, corresponding to a refuse production of 0.07 tonne, per person, per year. Refuse produced, by the black and white population, and landfilled, in Grahamstown, is thus the sum of 4 200 and 5 660 tonne, 11 950 tonne. Over a period of 5 years, then, the site should contain some 49 000 tonne.

Different estimates of the quantity of refuse in the Grahamstown Landfill Site are not the same (Table 21). The amount of wet-as-received refuse estimated from the refuse survey (88 000 tonne) is considerably higher than that calculated to be produced by the population (49 000 tonne) and higher than that calculated from site dimensions (62 000 tonne**). This could be due to a number of reasons:

- No account has been taken of a possible annual growth rate of waste production. It has been assumed that quantities of waste deposited in the first four years are the same as those presently deposited (1990/1991). Quantities deposited in earlier years may have been significantly lower.

**This estimate will obviously not be for refuse in the wet-as-received state, as the refuse may lose or gain water, depending on site conditions. It cannot therefore be strictly compared to data obtained from the refuse survey.

- When calculating quantities of landfilled refuse from the site volume, a compaction of 0.7 tonne.m^{-3} was assumed. No compaction measurements were taken. Compaction is possibly higher or lower.
- Waste production per capita may be higher.

6.2 Potential for Recycling

Considering that 13% of the refuse deposited (8.6 tonne.day^{-1}) consists of paper (Table 13), a total of 2 300 tonne of paper is deposited every year (190 tonne.$month^{-1}$). According to data supplied by Grahamstown Municipality, an average of 27 tonne per month of paper and cardboard was collected from the refuse, during the survey period (Table 15), for recycling by Ciskei Waste Collectors***. The fraction of paper removed for recycling thus amounts to only 14% (ww) of the paper deposited at the site (Table 17), and most of the paper is in fact landfilled.

Plastic was removed from the refuse for recycling for four months during the survey period. For these four months, an average of 1.5 tonne of plastic was recycled per month which is 3% of the plastic that is disposed of. Glass has recently been recycled and amounts to 1.8 tonne per month which is a mere 2% of the glass disposed of (Tables 15, 17).

The quantities of potentially recyclable materials that are

***During the survey period (July 1990-June 1991) no paper recycling was undertaken for a period of 5 months. The average quantity recycled per month has been calculated over 7 months.

landfilled, every year, are as follows:

paper.....2 300 tonne
glass.....1 100 tonne
metal.....1 100 tonne
plastic.....700 tonne

6.3 Quantity of Biodegradable MSW

The fraction of landfilled MSW that is biodegradable was calculated on the following grounds:

- The paper, food, garden and fines fractions were classified as biodegradable.
- Only 86% of the paper fraction was considered, as 14% is recycled.
- Analysis was on a wet mass basis.
- Builders rubble and soil were included in the analysis.

It is seen from Figure 2 (Table 18) that under moist conditions in the landfill, 48% (ww) of landfilled MSW is biodegradable. This amounts to 42 000 tonne of wet-as-received biodegradable material in the site. If the inherent moisture contents of the components are considered, only 36% (dw) of MSW is dry biodegradable material (Figure 3, Table 19). This is equivalent to a deposition rate of 4 400 tonne of dry biodegradable material per year (Figure 4), indicating that after five years the amount of dry biodegradable material in the site is 22 000 tonne.

6.4 Quantity of Biodegradable Domestic Refuse

Under moist landfill conditions, i.e. assuming that paper decomposes, 53% (dw), of dry domestic refuse is biodegradable (Table 7). Of the 6 500 tonne of domestic refuse (moisture content 30-35%) deposited at the site per year, 2 200 tonne is therefore dry biodegradable material.

7. SUMMARY

This survey set out to answer a number of questions (section 2):

(i) Quantity of MSW

Quantities of MSW are as follows:

	Deposited Per Year (1990-1991) tonne	Estimate of Total Mass in Site tonne
wet-as-received MSW	18 000	88 000
dry biodegradable MSW	4 400	22 000

(ii) Composition of domestic MSW

From Figure 5 (Table 5), paper (26%) and food (27%) appear to be the major components of wet-as-received domestic refuse. If moisture contents are considered, and analysis is on a dry basis, food only comprises 15% while paper comprises 30% (Figure 6, Table 7). The density of uncompacted domestic waste is on average 0.1 tonne.m^{-3} (on average, the 1m^3 sampling box contained 100 kg

refuse).

(iii) Composition of MSW

Domestic refuse makes up most of the deposited refuse, 37% (25 tonne.day^{-1}), followed by builders rubble, 28% (19 tonne.day^{-1}) and garden waste, 22% (15 tonne.day^{-1}). (Figure 7, Table 12). If the domestic refuse fraction is divided into its various components, builders rubble is by far the greatest component of dry MSW (37%, Figure 3).

(iv) Method of disposal

(a) Virtually all refuse that arrives at the site is landfilled, 98.8%.

(b) Paper (27 tonne.month^{-1}), glass (2 tonne.month^{-1}) and plastic (2 tonne.month^{-1}) is removed for recycling by Ciskei Waste Collectors.

(c) Quantities of recyclable materials that are landfilled, tonne.month^{-1}, are as follows:

paper.....190
glass.....90
metal.....90
plastic.....60

(d) None of the waste is incinerated.

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TABLE 1
Domestic Refuse Composition
Spring

COMPONENT	WET WEIGHT kg.m^{-3} Mon	Tues	Wed	Thurs	Fri	**Av.**	WET FRACTION weight %	DRY WEIGHT kg.m^{-3}	DRY FRACTION weight %
paper	34	33	39	32	51	**35**	31	27	35
plastic	10	9	11	12	21	**13**	11	11	14
food	21	36	49	40	19	**33**	28	12	17
glass	4	16	27	20	11	**16**	13	15	18
metal	3	7	6	9	7	**6**	5	6	7
fines	9	28	8	11	7	**8**	11	7	9
misc.	1	1	1	1	1	**1**	1	1	1
total	83	129	130	125	116	**116**	100	80	101

NOTE: • This survey was conducted during the period 22, 23, 24, 25 and 26 October 1990.
• The dry weight in column 4 is obtained using the moisture values for refuse components given in Table 6.

TABLE 2
Domestic Refuse Composition
Summer

COMPONENT	WET WEIGHT kg.m^{-3} Mon	Tues	Wed	Thurs	**Av.**	WET FRACTION weight %	DRY WEIGHT kg.m^{-3}	DRY FRACTION weight %
paper	24	28	28	31	**28**	30	21	34
plastic	10	12	8	8	**9**	10	8	13
food	16	26	29	22	**23**	24	9	13
glass	12	22	23	8	**16**	16	16	23
metal	3	8	5	6	**6**	6	5	8
fines	3	6	3	1	**3**	3	3	4
misc.	13	12	5	11	**10**	11	4	6
total	80	114	101	84	**95**	100	66	101

NOTE: • This survey was conducted during the period 3, 6, 18 and 19 December 1990.
• The dry weight in column 4 is obtained using the moisture values for refuse components given in Table 6.

TABLE 3
Domestic Refuse Composition
Autumn

COMPONENT	WET WEIGHT kg.m^{-3} Mon	Tues	Wed	Thurs	Fri	**Av.**	WET FRACTION weight %	DRY WEIGHT kg.m^{-3}	DRY FRACTION weight %
paper	22	30	21	17	21	**21**	23	17	26
plastic	11	9	11	12	12	**11**	11	9	15
food	29	24	21	35	11	**24**	25	9	14
glass	17	11	21	29	13	**18**	19	18	27
metal	4	5	5	6	5	**5**	5	5	7
fines	18	9	22	14	9	**14**	15	5	8
misc.	0	0	4	4	2	**2**	2	2	3
total	100	88	101	115	72	**95**	100	64	100

NOTE: • This survey was conducted during the period 20, 21, 22, 25 and 26 March 1991.
• The dry weight in column 4 is obtained using the moisture values for refuse components given in Table 6.

TABLE 4
Domestic Refuse Composition
Winter

COMPONENT	WET WEIGHT kg.m^{-3} Mon	Tues	Wed	Thurs	**Av.**	WET FRACTION weight %	DRY WEIGHT kg.m^{-3}	DRY FRACTION weight %
paper	22	29	17	25	**23**	21	18	25
plastic	20	14	7	7	**12**	11	10	14
food	45	23	44	19	**33**	29	12	17
glass	24	12	13	18	**17**	15	16	22
metal	6	3	5	8	**5**	5	5	7
fines	17	19	20	10	**17**	15	6	9
misc.	18	0	0	0	**5**	5	5	7
total	151	99	105	85	**110**	101	72	101

NOTE: • This survey was conducted during the period 12, 13, 17 and 18 June 1991.
• The dry weight in column 4 is obtained using the moisture values for refuse components given in Table 6.

TABLE 5
Analysis of Domestic Refuse Composition
Wet Weight Basis

COMPONENT	FRACTION / wet weight %				
	Spring	Summer	Autumn	Winter	**Average**
paper	31	30	23	21	**26**
plastic	11	10	11	11	**11**
food	28	24	25	29	**27**
glass	13	16	19	15	**16**
metal	5	6	5	5	**5**
fines	11	3	15	15	**11**
misc.	1	11	2	5	**5**
total	100	100	100	101	101

TABLE 6
Moisture Contents of Components of MSW
(After Vesiland and Pereira[6])

COMPONENT	MOISTURE / %
paper	23
glass	3
metal	5.5
plastic	13
food	63
misc.	4
fines†	63

NOTE: † Assumed by the author. The fines fraction is that fraction that is left once the readily identifiable fractions have been removed. It consists mostly of food and paper.

TABLE 7
Analysis of Domestic Refuse
Dry Weight Basis

COMPONENT	FRACTION / dry weight %				
	Spring	Summer	Autumn	Winter	Average
paper	35	34	26	25	30
plastic	14	13	15	14	14
food	17	13	14	17	15
glass	18	23	27	22	23
metal	7	8	7	7	7
fines	9	4	8	9	8
misc.	1	6	3	7	4
total	101	101	100	101	101

TABLE 8
MSW Composition
Spring

COMPONENT	WEIGHT tonne.day^{-1}					FRACTION wet weight %
	Tues	Wed	Thurs	Fri	**Av.**	
domestic	25	27	31	24	**27**	33
garden	11	14	17	18	**15**	18
rubble	2	99	14	18	**18**	40
metal	0	6	1	1	**2**	2
paper	1	3	0	4	**2**	2
glass	0	1	1	0	**1**	1
abattoir	1	1	2	2	**2**	2
light.ind.	1	0	0	1	**1**	1
total	41	151	66	68	**83**	99

NOTE:
- This survey was conducted during the period 23, 24, 25 and 26 October 1990.
- The fraction of the waste labelled "rubble" is builders rubble and includes soil.
- The fraction of the waste labelled "light. ind." is light industrial waste which consists largely of tyres.

TABLE 9
MSW Composition
Summer

COMPONENT	WEIGHT tonne.day^{-1}			FRACTION wet weight %
	Tues	Thurs	Av.	
domestic	18	32	25	45
garden	9	7	8	15
rubble	3	7	5	9
metal	13	2	8	14
paper	2	8	5	9
glass	1	0	1	1
abattoir	2	2	2	4
light.ind.	2	2	2	4
total	48	57	54	101

NOTE:
- This survey was conducted during the 6 and 18 December 1990.
- The fraction of the waste labelled "rubble" is builders rubble and includes soil.
- The fraction of the waste labelled "light. ind." is light industrial waste which consists largely of tyres.

TABLE 10
MSW Composition
Autumn

COMPONENT	WEIGHT tonne.day^{-1}				FRACTION wet weight %
	Wed	Thurs	Fri	Av.	
domestic	33	16	25	25	36
garden	23	26	22	23	34
rubble	15	18	17	17	24
metal	0	1	0	0	0
paper	0	0	0	0	0
glass	0	0	0	0	0
abattoir	3	2	3	3	4
light.ind.	0	1	2	1	2
total	74	64	69	69	100

NOTE:
- This survey was conducted during the period 20, 21 and 22 March 1991.
- The fraction of the waste labelled "rubble" is builders rubble and includes soil.
- The fraction of the waste labelled "light. ind." is light industrial waste which consists largely of tyres.

TABLE 11
MSW Composition
Winter

COMPONENT	WEIGHT tonne.day^{-1}						FRACTION wet weight %
	Mon	Tues	Wed	Thurs	Fri	Av.	
domestic	33	21	12	23	27	**23**	35
garden	7	14	9	24	13	**13**	20
rubble	5	15	15	37	18	**22**	34
metal	1	0	4	1	0	**1**	2
paper	2	0	0	1	0	**1**	2
glass	0	0	0	0	0	**0**	0
abattoir	5	5	0	16	0	**5**	8
light.ind.	1	0	0	1	0	**0**	0
total	54	55	39	101	58	**65**	101

NOTE:
- This survey was conducted during the period 12, 13, 14, 17 and 18 June 1991.
- The fraction of the waste labelled "rubble" is builders rubble and includes soil.
- The fraction of the waste labelled "light. ind." is light industrial waste which consists largely of tyres.

TABLE 12
Analysis of MSW Composition
Wet Weight Basis

COMPONENT	WEIGHT tonne.day^{-1}					FRACTION wet weight %
	Spring	Summer	Autumn	Winter	**Average**	
domestic	27	25	25	23	**25**	37
garden	15	8	23	13	**15**	22
rubble	33	5	17	22	**19**	28
metal	2	8	0	1	**3**	4
paper	2	5	0	1	**2**	3
glass	1	0	0	0	**0**	0
abattoir	2	2	3	5	**3**	4
light.ind.	1	2	1	0	**1**	1
total	83	54	69	65	**68**	99

NOTE: · The fraction of the waste labelled "rubble" is builders rubble and includes soil.
· The fraction of the waste labelled "light. ind." is light industrial waste which consists largely of tyres.

TABLE 13
Complete Analysis of MSW Composition
Wet Weight Basis

COMPONENT	WET WEIGHT tonne.day^{-1}					FRACTION wet weight %
	Spring	Summer	Autumn	Winter	Average	
paper	10	13	6	6	**9**	13
plastic	3	3	3	3	**3**	4
food	8	6	6	7	**7**	10
garden	15	8	23	13	**15**	22
fines	3	1	4	4	**3**	4
glass	5	4	5	4	**4**	6
metal	3	10	1	2	**4**	6
misc.	0	3	1	1	**1**	2
rubble	33	5	17	22	**19**	28
abattoir	2	2	3	5	**3**	4
light.ind.	1	2	1	0	**1**	1
total	83	54	69	65	**68**	100

NOTE: • The values in Table 13 are obtained fron data in Tables 1, 2, 3, 4 and Table 12.

TABLE 14
Different Surveys of MSW Composition / wet weight %

FRACTION	UK[2]	USA[5]	WORLD [1] (i)	WORLD [1] (ii)	SOUTH AFRICA RNB[4]	SOUTH AFRICA PMB[4]	SOUTH AFRICA GHT
paper†	29	31	1-10	15-40	11	28	13
plastic	7	8	1-5	2-6	4	10	4
food†	-	-	-	-	-	-	10
putresc†	19	-	-	-	-	-	-
garden†	6	-	-	-	15	4	22
organic†	-	-	-	-	6	32	-
veg. matter†	-	24	40-85	20-65	-	-	-
fines†	9	-	-	-	-	-	4
glass	8	11	1-10	1-10	4	10	6
metal	9	8	1-5	1-5	5	7	6
misc.	13	9	1-40	1-30	1	9	7
rubble + dust	-	9	-	-	54	0	28
BIODEGRADABLE/ %	63	55	-	-	32	64	48

NOTE:
- column 2: United Kingdom, reference 2
- column 3: United States of America, reference 5
- column 4: (i) low income population, reference 1
  (ii) middle income population, reference 1
- column 5: RNB: Randburg, reference 4
  PMB: Pietermaritzburg, reference 4
  GHT: Grahamstown, this survey

† Fractions assumed to be biodegradable

TABLE 15
Materials Recycled
July 1990 - June 1991

MONTH	WEIGHT RECYCLED / kg.month^{-1}				
	Paper	Cardboard	Plastic	Glass	TOTAL
July '90	13948	8020	3768	0	25736
Aug '90	18589	11397	1033	0	31019
Sep '90	21370	12302	0	0	33672
Oct '90	19088	11005	0	0	30093
Nov '90	30006	8852	0	0	38858
Dec '90	0	0	0	0	0
Jan '91	0	0	0	0	0
Feb '91	0	0	0	0	0
March '91	0	0	0	0	0
April '91	0	0	0	0	0
May '91	9585	4983	1284	1613	17465
June '91	10960	6196	1028	1921	20105
TOTAL	123546	62755	7113	3534	196948

NOTE: Data supplied by Grahamstown Municipality.

TABLE 16
Materials Recycled Seasonally

COMPONENT	WEIGHT RECYCLED / kg.day^{-1}			
	Spring	Summer	Autumn	Winter
paper	1 505	0	0	858
plastic	0	0	0	51
glass	0	0	0	96
TOTAL	1 505	0	0	1 005

NOTE: • The "paper" component in column 1 includes cardboard.
• The values in this table are obtained from data in Table 14.

TABLE 17
Fractions of MSW Recycled Seasonally

COMPONENT	FRACTION OF REFUSE COMPONENT RECYCLED / %			
	Spring	Summer	Autumn	Winter
paper	14	0	0	15
plastic	0	0	0	2
glass	0	0	0	3
MSW	2	0	0	2

NOTE: The values in this table are obtained from data in Tables 13 and 15.

TABLE 18
Landfilled MSW Composition
Wet Weight Basis

COMPONENT	WEIGHT tonne.day^{-1}					FRACTION wet weight %
	Spring	Summer	Autumn	Winter	Average	
paper	9	13	6	5	8	12
plastic	3	3	3	3	2	4
food	8	6	6	7	7	10
garden	15	8	23	13	15	22
fines	3	1	4	4	3	4
glass	5	4	5	3	4	6
metal	3	10	1	2	4	6
misc.	0	3	1	1	1	2
rubble	33	5	17	22	19	28
abattoir	2	2	3	5	3	4
light.ind	1	2	1	0	1	1
TOTAL	82	55	69	61	68	99

NOTE: The values in this table are obtained from data in Tables 13 and 15.

TABLE 19
Landfilled MSW Composition
Dry Weight Basis

COMPONENT	WET WEIGHT tonne.year^{-1}	MOISTURE %	DRY WEIGHT tonne.year^{-1}	FRACTION dry weight %
paper	2 080	23	1 600	13
plastic	700	13	610	5
food	1 740	63	644	5
garden	3 850	50	1 930	16
fines	730	63	270	2
glass	1 090	3	1 060	9
metal	1 070	5.5	1 010	8
misc.	310	4	300	2
rubble	5 020	10	4 520	37
abattoir	780	95	40	0
light.ind	260	5	250	2
TOTAL	17 630		12 230	99

NOTE:
- The values in column 2 are obtained from data in Table 17; 1 year = 260 working days.
- Values in column 3 are those in Table 5.
- The "garden" component in column 1 consists of approximately 50% wood and 50% grass, leaves etc.

TABLE 20
The Calculation of Refuse Quantities using Site Dimensions

REGION	AREA m^2	DEPTH m	VOLUME m^3	COMPACTION tonne.m^{-3}	WEIGHT tonne
A	650	10	6 500	0.7	4 550
B	2 120	20	42 400	0.7	29 700
C	1 300	20	26 000	0.7	18 200
D	470	20	9 400	0.7	6 600
E	400	10	4 000	0.7	2 800
TOTAL	5 000		88 000		62 000

NOTE: The regions A, B, C, D and E are those so indicated in Figure 1.

TABLE 21
Estimates of MSW Quantities

SOURCE	PRODUCTION tonne.yr^{-1}		LANDFILLED tonne.5yr^{-1}	
	wet weight	dry weight	wet	dry
refuse survey	18 000	12 000	88 000	61 000
site dimensions	-	-	62 000	
population	5 660+4 200		49 000	

NOTE:
- The values in the third column represent the amount of refuse in the Grahamstown Landfill.
- The estimate based on site dimensions cannot be given on wet or dry weight terms as it is not known how much water the refuse has lost/gained.

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FIGURE	DESCRIPTION
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2	Landfilled MSW: Wet Weight %
3	Landfilled MSW: Dry Weight %
4	Landfilled MSW: Dry Tonne.Yr^{-1}
5	Domestic Waste Composition: Wet Weight %
6	Domestic Waste Composition : Dry Weight %
7	MSW Composition: Wet Weight %

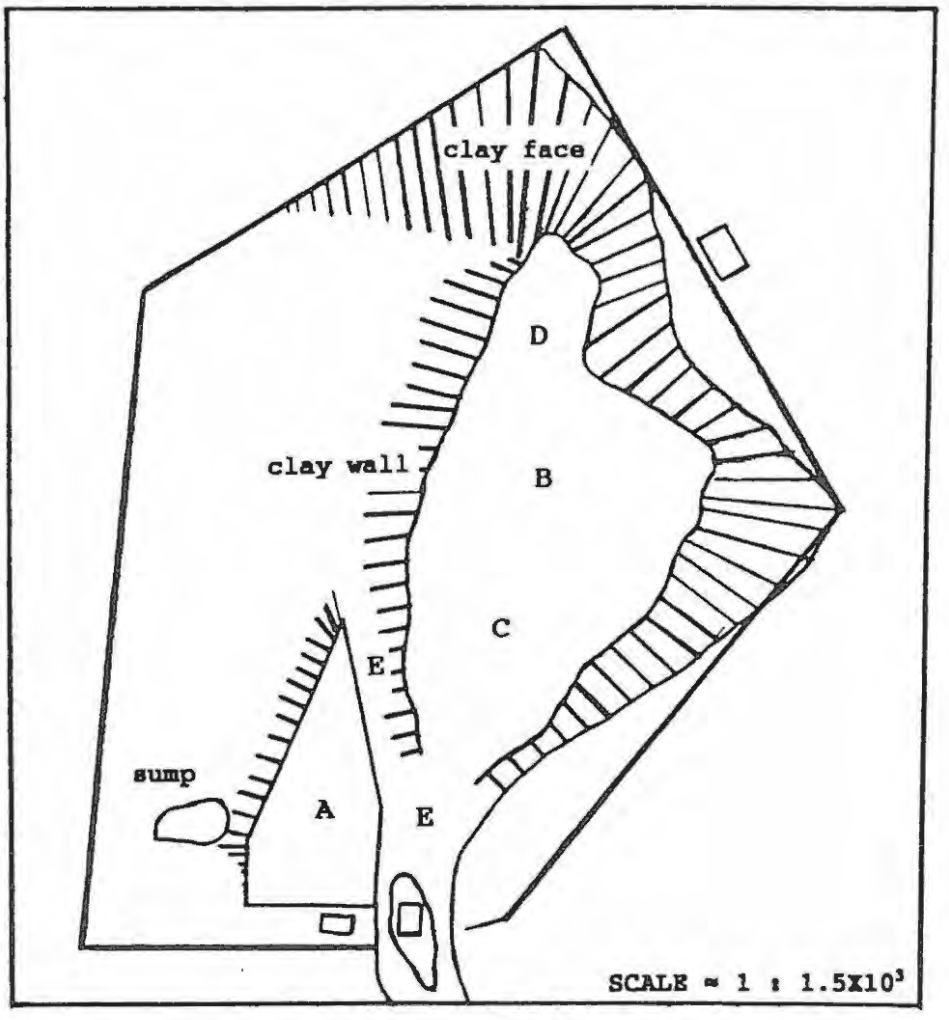

Figure 1. Schematic Diagram of Grahamstown Landfill (Repeated from Chapter 4)

A = rehabilitated site
B = upper tipping site
C = lower tipping site
D = upmost tipping site
E = edges and roads

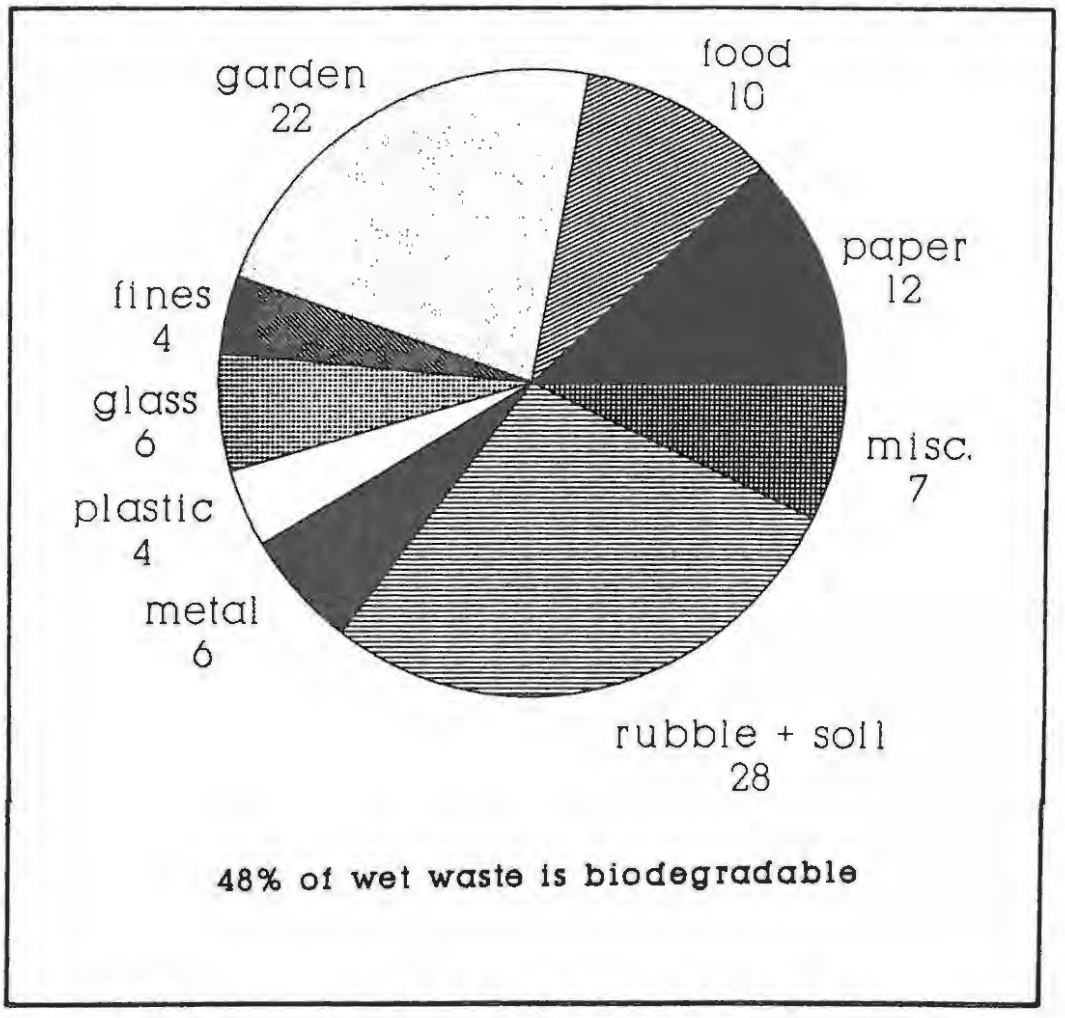

Figure 2. Landfilled MSW: Wet Weight %
(Data taken from Table 18)

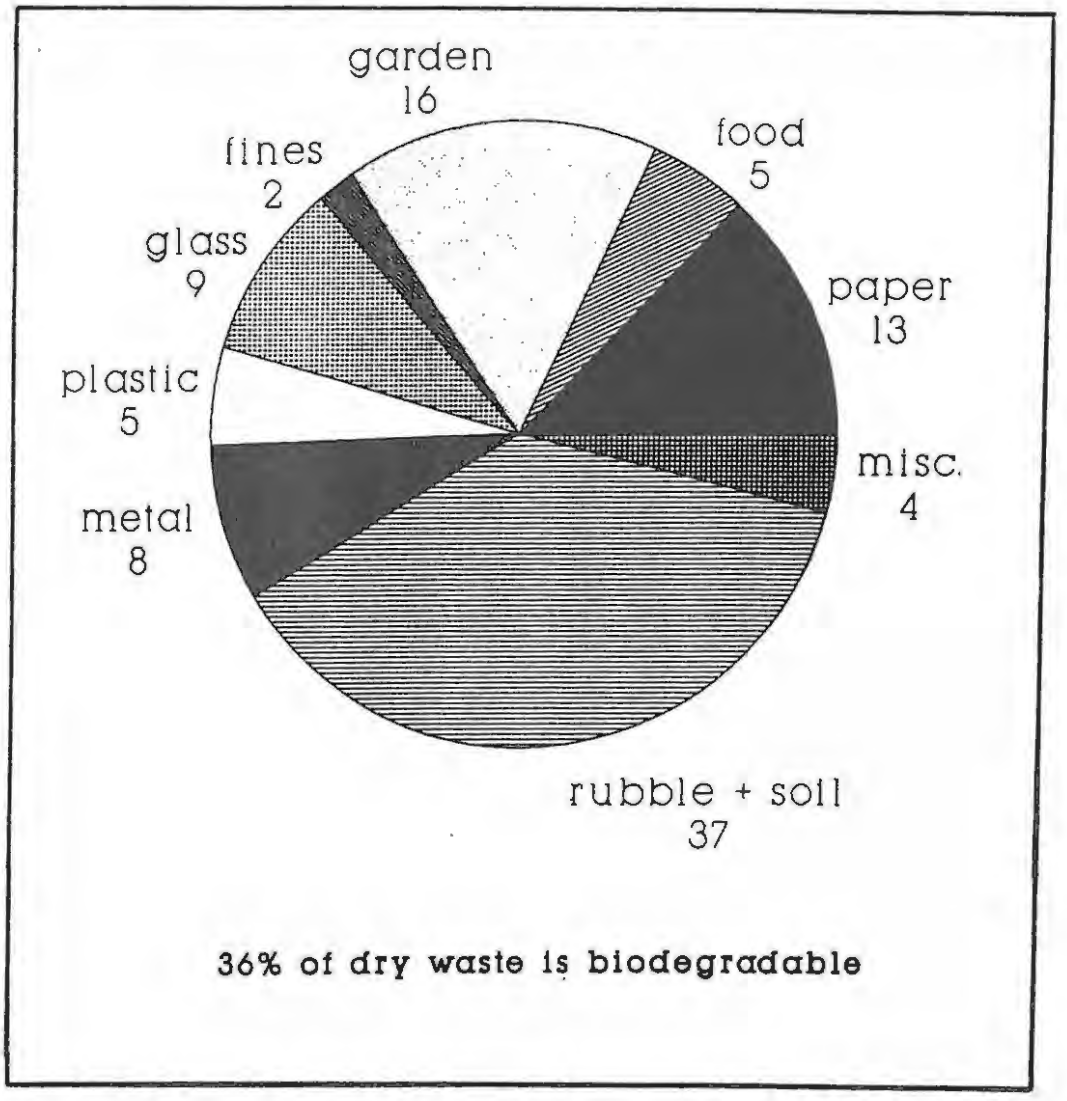

Figure 3. Landfilled MSW: Dry Weight %
(Data taken from Table 19)

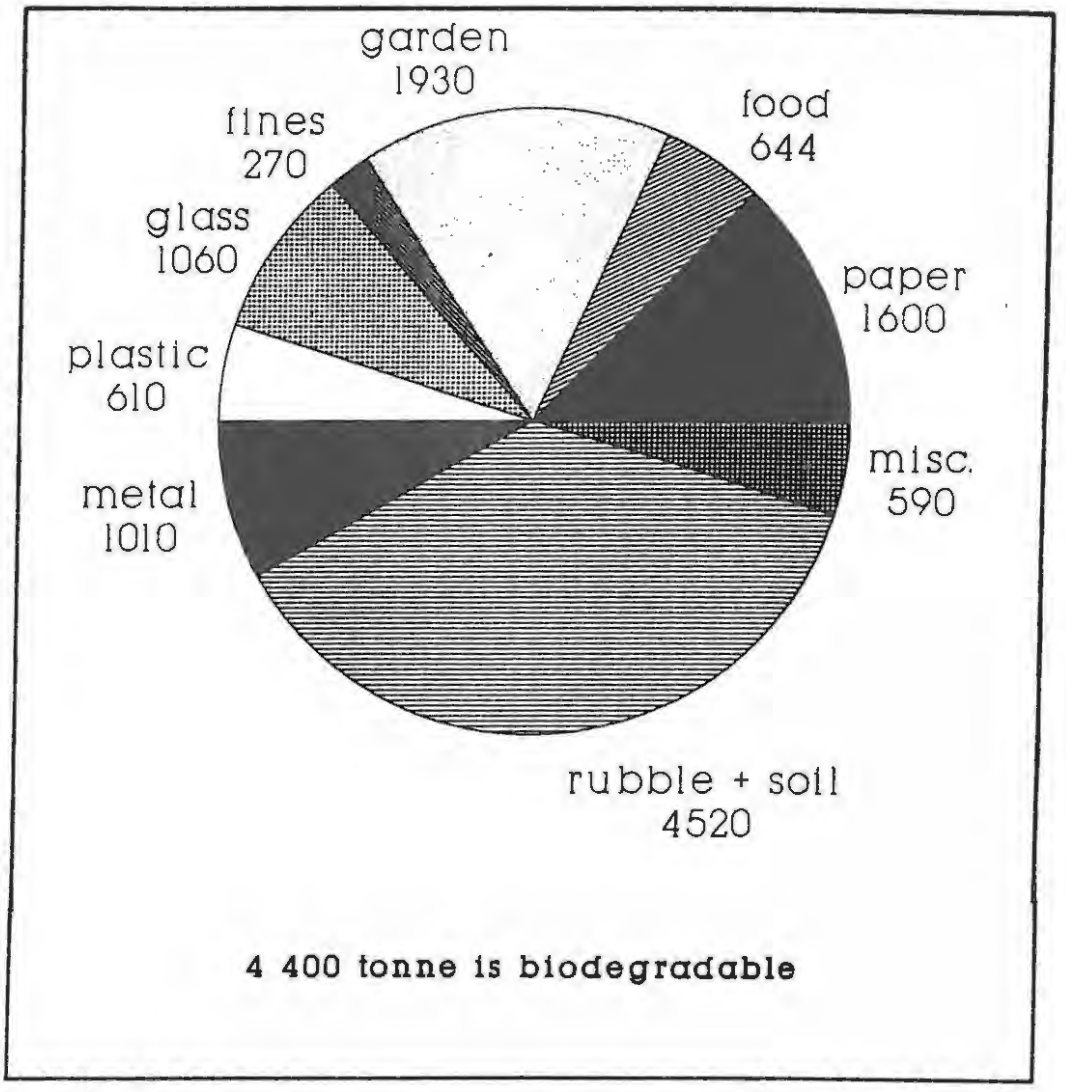

Figure 4. Landfilled MSW: Dry Tonne.Yr^{-1}
(Data taken from Table 19)

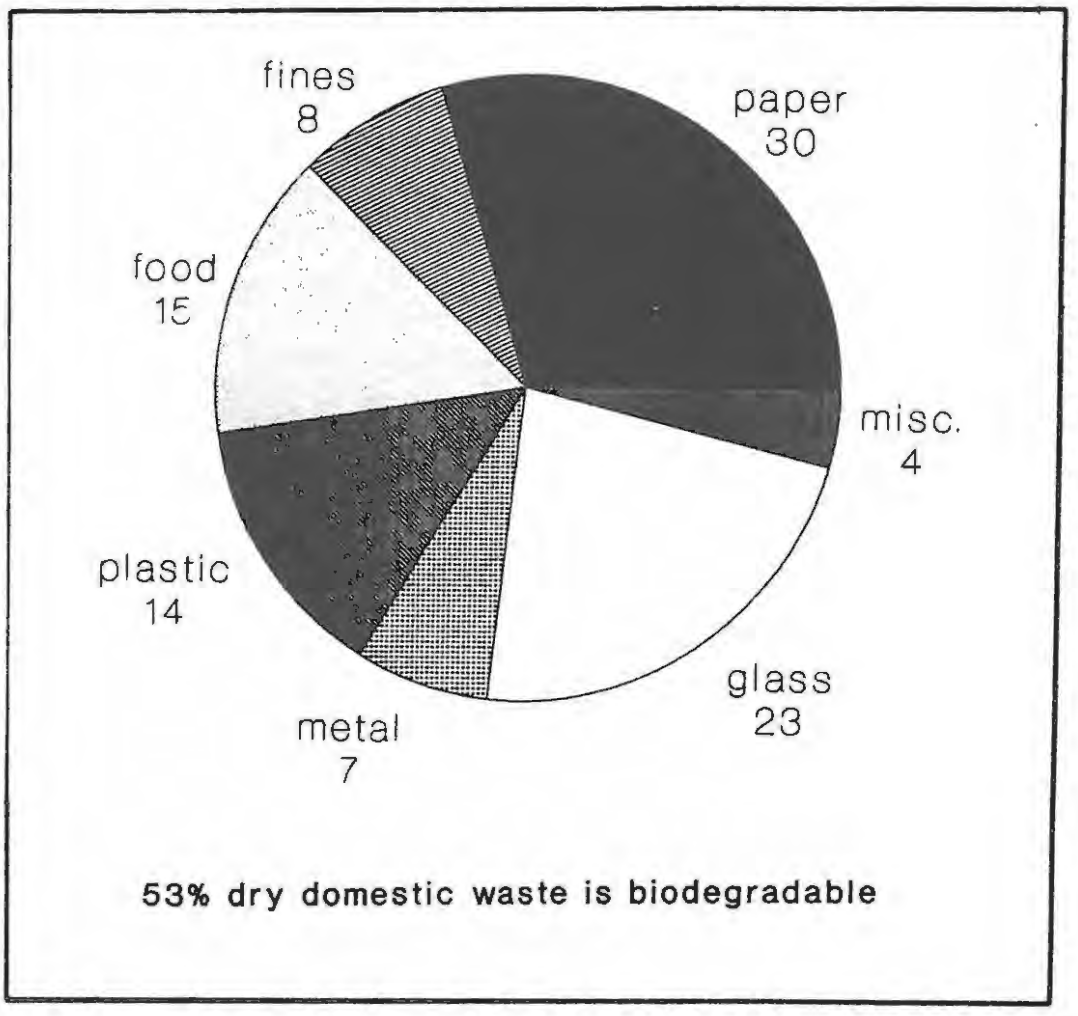

Figure 6. Domestic Waste Composition: Dry Weight %
(Data taken from Table 7)

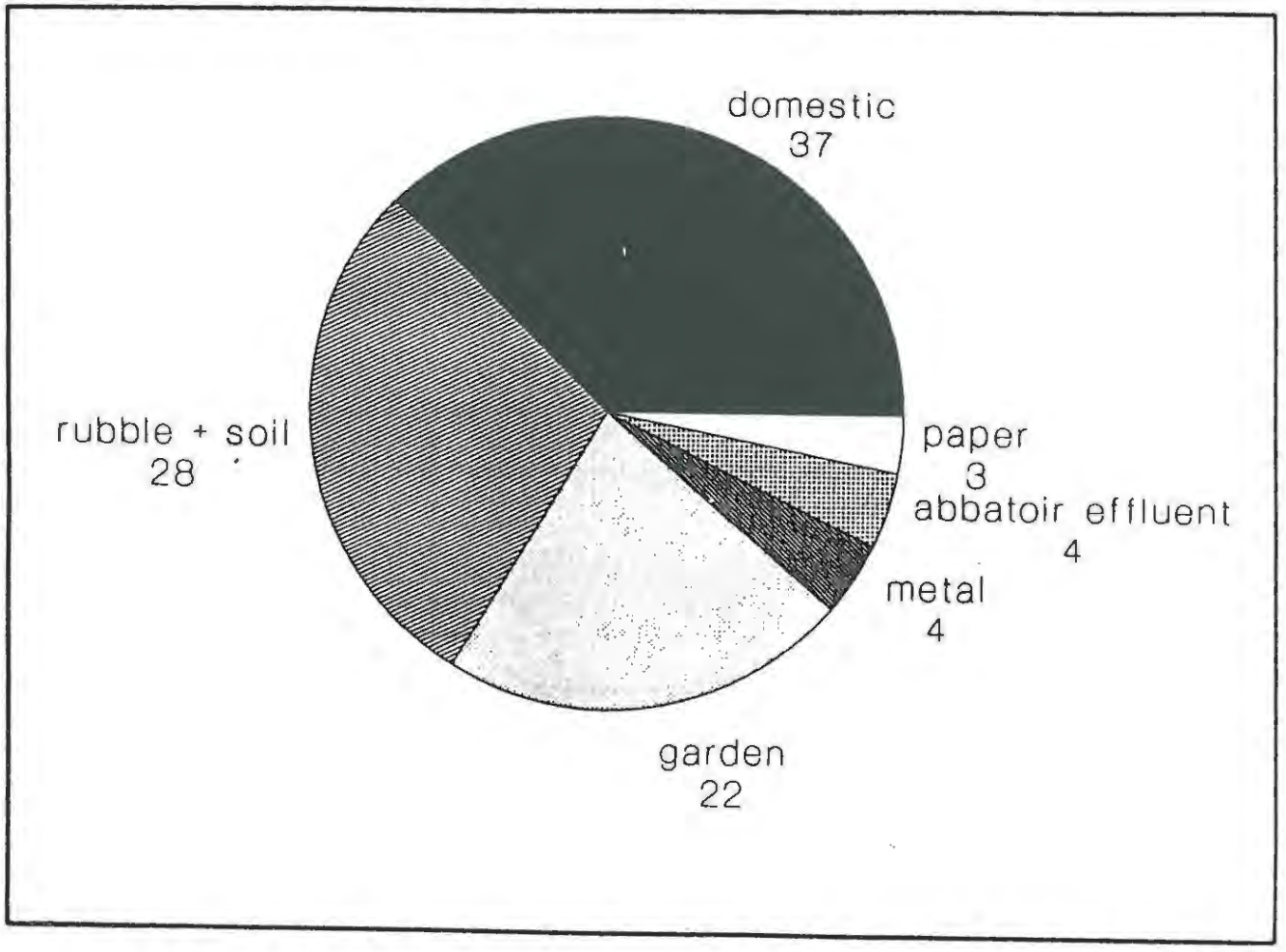

Figure 7. MSW Composition: Wet Mass %

CHAPTER 9

MODELLING OF LANDFILL GAS GENERATION

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3. MODELS

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1. INTRODUCTION

In order to assess the feasibility of any proposed landfill gas exploitation scheme, determination of the potential quantity of gas available from the site and the rate at which this gas is produced is necessary. Both of these parameters can be derived from the kinetic modelling of the LFG generation process.

Mathematical LFG generation models were first developed in the United States in the late 1970's in response to needs to extract LFG for commercial ends. New models and improvements on previous models have since been made, despite the difficulties encountered in modelling a multivariant system such as a landfill. As a result of the difficulties encountered, all models are based on certain assumptions and only on very limited data. None of the models have been clearly verified by field data[3,9]. Despite this, the use of models is becoming more important as more landfills are exploited for methane, and more data on methane flow rates becomes available.

2. KINETICS OF REFUSE DECAY

The classical Monod equation (cited EMCON Associates, 1983[3]) relating rate of substrate utilisation to both the concentration of micro-organisms in the system and to the concentration of soluble substrate surrounding the organisms, is a first-order rate equation. If one considers that in a landfill the limiting nutrient for the methanogens is the biodegradable organic waste,

rate constants of the degradation reactions. The author has chosen not to discuss the models in chronological order.

3.1 Modified Hoeks Model (MH Model)[7]

Refuse does not begin to produce methane as soon as it is landfilled, but undergoes a first stage of aerobic decay (during which carbon dioxide is produced), before anaerobic conditions set in and methane is produced. Methane is then produced in increasing quantities until a stage of maximum production is reached[4]. This theoretical model neglects the initial stage of aerobic decay and that of increasing gas production as it is assumed that this stage is very short. It is assumed in the model that the production of LFG therefore takes place in one stage, starting at a maximum, and decreasing with time. Development of the model, as interpreted and slightly modified by the author, is given below.

Firstly, it is assumed that first-order kinetics govern the rate of refuse decay (equation (1)). Integration of equation (1) gives rise to an exponential term:

$$P_t = P_0 \exp(-kt) \quad \ldots\ldots (2)$$

where: P_o = quantity of degradable material at time t = 0

Differentiating equation (2) with respect to time yields:

$$\frac{dP_t}{dt} = -kP_0 \exp(-kt) \ldots\ldots (3)$$

In order to relate P_t to gas production, it is necessary to consider the overall reaction for the production of LFG, which may be described by:

$$2(CH_2O)_n \rightarrow nCH_4 + nCO_2 \ldots\ldots (4)$$

where: $(CH_2O)_n$ is glucose and represents the biodegradable fraction of refuse.

It is readily calculated that a maximum of 416 m^3 of methane can be produced for every tonne of biodegradable material that degrades:

2 moles $(CH_2O)_n$	→	1 mole CH_4
60 g CH_2O	→	25 l CH_4
∴ 1000 kg CH_2O	→	416 m^3 CH_4

where: the molar volume of methane is 25 litre at STP

Assuming that gas production is dependant on the decomposition of organic material (a reasonable assumption):

$$\alpha dP_t = -d[CH_4] \ldots\ldots (5)$$

where: $[CH_4]$ is the quantity of methane produced (m^3)
α is the maximum amount of methane that can be obtained from refuse, as calculated above (416 $m^3.tonne^{-1}$)

Using equations (3) and (5):

$$\frac{d[CH_4]}{dt} = \alpha k P_0 \exp(-kt) \ldots\ldots (6)$$

where: k, the rate constant, is dependant on the nature of the organic material and is a function of its half-life ($t_{1/2}$):

$$k = \frac{ln2}{t_{1/2}} \ldots\ldots (7)$$

Substituting for α and k, the rate equation becomes:

$$\frac{d[CH_4]}{dt} = \frac{416 x 0.693}{t_{1/2}} x P_0 \exp\left(\frac{-0.693 x t}{t_{1/2}}\right) \ldots\ldots (8)$$

The potential maximum methane production (in $m^3.hr^{-1}$) can hence be calculated over the lifetime of a landfill, provided that the amount of dry biodegradable refuse in the site (P_o) and the half-life ($t_{1/2}$) of the refuse is known. The total rate of methane production is in fact the sum of the methane produced from individual refuse components, which differ in their susceptibility to decay. Equation (8) hence becomes:

$$\frac{d[CH_4]}{dt} = \sum_{j=1}^{j=3} \sum_{t=1}^{t=t} \frac{416 x 0.693}{t_{1/2(j)}} x P_{0(j)} \exp\left(\frac{-0.693 x t}{t_{1/2(j)}}\right) \ldots\ldots (9)$$

where: j refers to the various refuse components with particular half-lives.

This model classifies degradable refuse into three fractions according to the susceptibility to decay and assigns assumed $t_{1/2}$

values to these fractions as shown in Table 1.

TABLE 1
Classification of Biodegradable Refuse

DEGRADABILITY	$t_{1/2}$ /yr
readily	1
moderately	5
slowly	15

The parameter $t_{1/2}$ is a very important one when quantifying gas generation rates as the model is fairly sensitive to changes in this parameter. Unfortunately, and understandably, data on rates of degradation are scarce, with lysimeter tests appearing to be of little quantitative value in predicting field rates[6,7,9]. Because the study of LFG generation is a relatively new area of research, data from gas extraction experiments in the field are minimal, although they are now being extensively collected[9]. Data are very site-specific, and are dependent on landfill conditions and on moisture levels in particular. For example, estimates of composite $t_{1/2}$ values for United States landfills are 10-25 years for those in dry areas, 5-10 years for medium precipitation areas and 2-5 years for wet precipitation areas[9]. Ideally, $t_{1/2}$ values should be determined for the particular site being modelled (from start to completion of waste degradation), so that the effect of variables such as moisture and temperature is automatically considered. This is, however, not practical and at present values based on literature data must be assumed.

Using equation (9), the model predicts a gas generation profile

of the type shown in Figure 1. It is seen that the generation rate starts at a maximum and decreases exponentially with time.

Site specific, rate-determining factors such as moisture and temperature effect the value of α, 416 $m^3.tonne^{-1}$ (theoretical maximum). Available experimental values of α are between 10% and 30% of this value; 40-125 $m^3.tonne^{-1}$[6,9].

3.2 Environmental Resources Limited Model (ERL Model)[8]

This model was developed in order to assess the LFG resource in England and Wales. The model, which is essentially theoretical, but is constrained by experimental data, differs from the MH Model in two respects. Firstly, the initial aerobic phase of degradation is considered in the ERL Model while this is not so in the MH Model. This has a very important impact on the amount of organic substrate available for the methanogenic bacteria. Secondly, the concept of "gas producing carbon" is used in the ERL Model, whereby only a fraction of the biodegradable material is considered to produce methane. The model is very sensitive to both these limitations, and as a consequence the predictions of gas production are far lower than those for the MH Model. Apart from these two factors, the models are essentially the same.

The rate equation for the production of methane gas from anaerobic refuse degradation is written as:

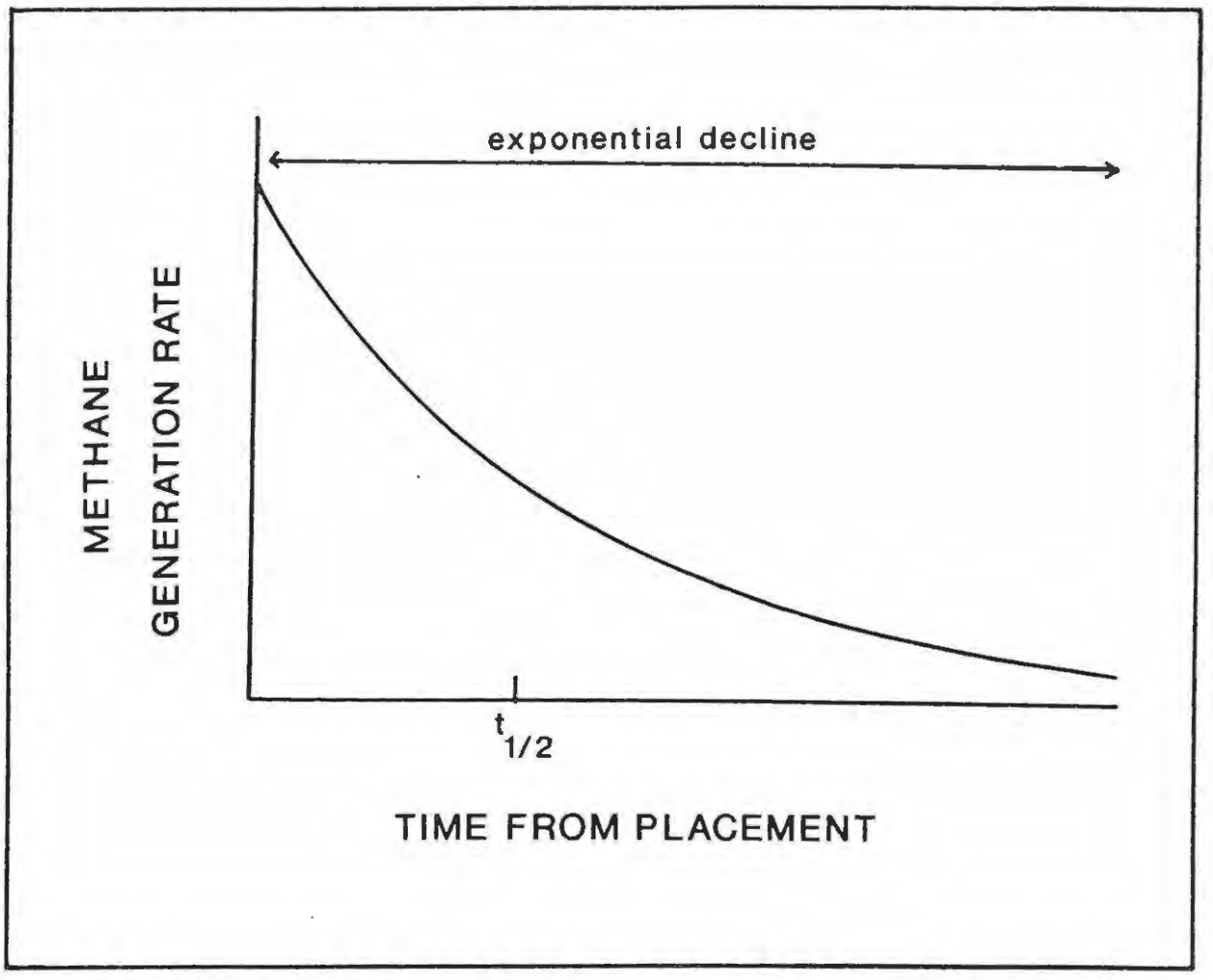

Figure 1. Gas Generation Profile as Predicted by Hoeks Model[7]

$$C_{CH_4}(t > t_a) = xC_{ao}\exp(-k_a t_a)\,[1-\exp(-k_n(t-t_a)]\;.....(10)$$

where: C_{CH4} is amount of methane*
t is time
t_a is the time at which the anaerobic reaction is complete
x is the stoichiometric coefficient (0.4-0.6)
C_{ao} is initial degradable carbon content
k_a is the rate constant for the aerobic reaction
k_n is the rate constant for the anaerobic reaction

Differentiation of equation (10) with respect to time yields an equation describing maximum methane production rate in terms of mass per unit time:

$$\frac{dC_{CH_4}}{dt} = -XC_{ao}\exp(-k_n t_a)\,[\exp(-k_n(t-t_a))]\,(-k_n)\;.....(11)$$

This equation can be written to express the rate of methane generation in terms of volume per unit time by introducing the parameter α (=416 m^3.tonne dry biodegradable $waste^{-1}$) to replace x, which is simply the methane content of the gas produced. The initial degradable carbon content, C_{ao}, is similarly replaced with the amount of biodegradable waste, P_o. Equation (11) then becomes:

$$\frac{d[CH_4]}{dt} = -\alpha P_0\exp(-k_a t_a)\,[\exp(-k_n(t-t_a))]\,(-k_n)\;.....(12)$$

OR

---

*The ERL Model defines the parameter C_{CH4} as the concentration of methane[8]. The parameter has units of mass however.

$$\frac{d[CH_4]}{dt} = \frac{416x0.693}{t_{1/2n}} xP_0 \exp\left(\frac{-0.693}{t_{1/2n}}(t-t_a)\right)\exp\left(\frac{-0.693}{t_{1/2a}}(t_a)\right) \ldots (13)$$

where: $t_{1/2n}$ is the half-life of the anaerobic reaction
$t_{1/2a}$ is the half-life of the aerobic reaction

This rate equation is now expressed in the same manner as that in the MH Model (equation (8)) and can be used to find the maximum rate of methane production with time (in $m^3.hr^{-1}$) provided that half-life values are known for the aerobic and anaerobic reactions. The model assumes that aerobic decay is of short duration, persisting for a period of 42 weeks, or 5 half-lives which are each 8 weeks long (i.e. $t_{1/2a}$ = 8 weeks). Half-lives for the anaerobic reaction are assigned in the same manner as for the MH Model.

The introduction of a term in the model which accounts for aerobic degradation of the waste, results in only 3% of the waste being available for methanogenesis[8]! This is somewhat startling, the indication being that models that do not take this phase into account drastically overestimate gas production.

The methane generation profile is the same (but much smaller in magnitude) as that for the MH Model, apart from the initial lag-time, during which aerobic decay takes place (Figure 2).

3.3 Palos Verdes Model (PV Model)[3]

This model was developed in the late 1970,s as part of the Palos Verdes Landfill Gas Development Project. It is a theoretical

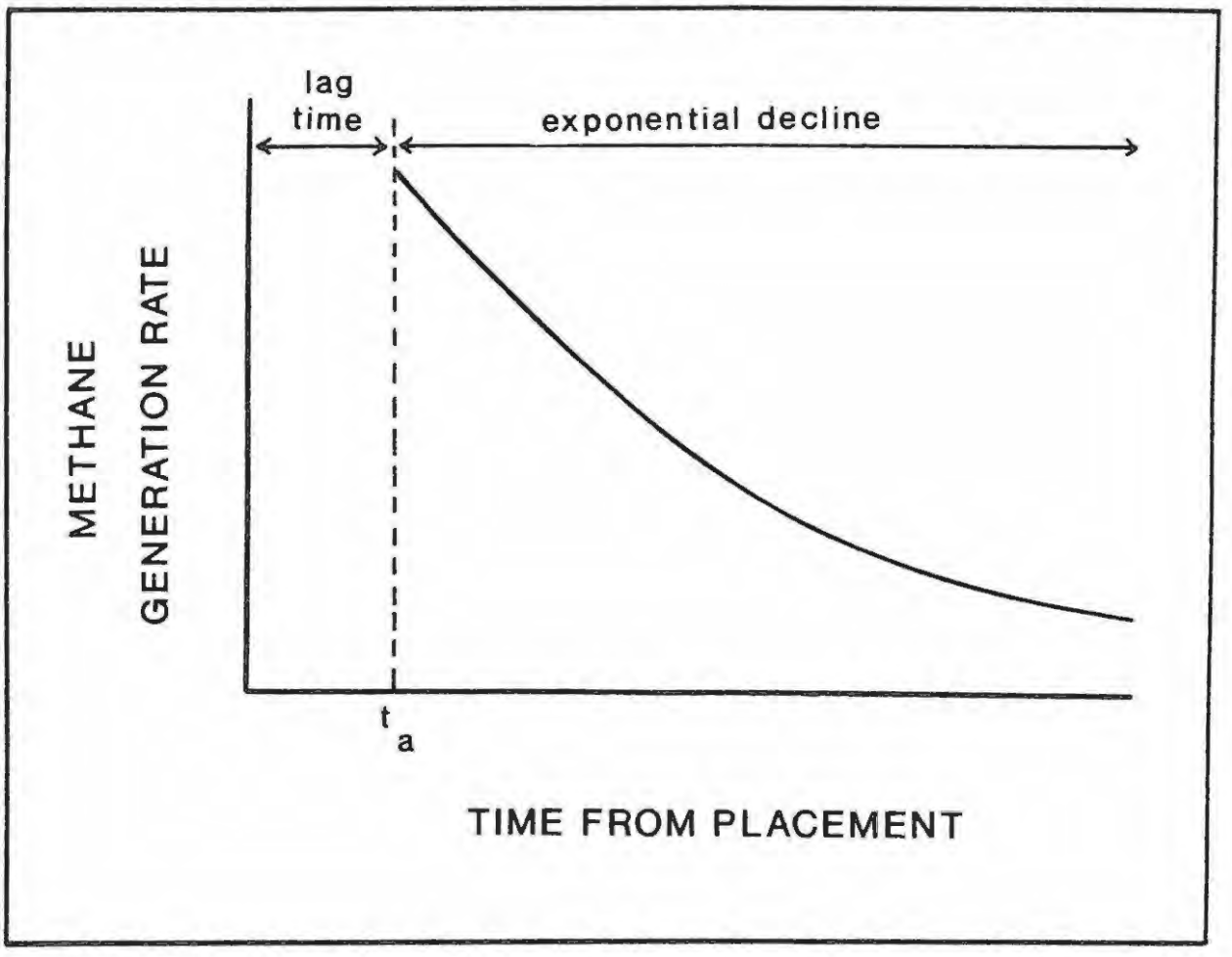

Figure 2. Gas Generation Profile as Predicted by Environmental Resources Limited Model[8]

first-order kinetic model which considers two stages of methane generation, unlike the MH and ERL Models which consider only a decreasing stage of methane generation.

It is assumed that during the first stage, methane generation rate is proportional to the volume of methane already produced. In the second stage, it is assumed that the rate of decrease in methane production is proportional to the volume of methane remaining to be produced.

Two simple first-order kinetic equations describe both stages; equation (14) describes the first stage and equation (15) the second:

$$\frac{dG}{dt} = k_1 G \quad \ldots\ldots (14)$$

$$\frac{dL}{dt} = -k_2 L \quad \ldots\ldots (15)$$

where: G = volume of gas produced prior to time t
L = volume of gas to be produced after time t
k_1 = first stage rate constant
k_2 = second stage rate constant

It is assumed that the maximum rate of gas production (which occurs at the transition from first stage to second stage kinetics) takes place when half of the ultimate gas production is reached (i.e. $G = L_o/2$ when $t = t_{1/2}$, where L_o is the total volume of gas to be produced). This is unlike the MH and ERL Models, in which maximum gas production occurs at time t = 0 in

the first case, and at $t = t_a$ (the time taken for anaerobic degradation) in the latter.

It is further assumed that the first-order kinetic equation (equation (14)) describing the first stage cannot be applied at inception of methane formation in a landfill. This restraint is necessary because one of the premises is that the production rate is proportional to the volume of gas already produced. The model therefore assumes that equation (14) can only be applied once 1% of the ultimate gas yield is produced (i.e. G_o, the volume of gas produced at time $t = 0$, is equal to $L_o/100$).

Considering the above two assumptions, integration of equation (14) leads to:

$$lnG = lnG_0 + k_1 t \quad \ldots\ldots (16)$$

Now, at $t = t_{1/2}$, $G = L_o/2$ and therefore:

$$G = \frac{L_0}{2} \exp(-k_1 (t_{1/2} - t)) \quad \ldots\ldots (17)$$

Integration of the second stage equation (equation (15)) yields:

$$lnL = \ln(\frac{L_0}{2}) - k_2 (t - t_{1/2}) \quad \ldots\ldots (18)$$

Now, at $t > t_{1/2}$, $G = L_o - L$, therefore:

$$G = L_0 [1 - (\frac{\exp(-k_2 (t - t_{1/2}))}{2})] \quad \ldots\ldots (19)$$

The model assumes that degradable waste can be classified according to one of two categories; readily decomposable (food and grass) and moderately decomposable (paper, wood, textiles etc.) waste. These fractions are assigned half-lives of 1 and 2 years respectively. The value of the first-stage rate constant (k_1) can then be calculated for each category of waste by rearranging equation (16) to give:

$$k_1 = \frac{\ln(\frac{G}{G_0})}{t} \quad \ldots\ldots (20)$$

At $t = t_{1/2}$, $G = L_0/2$ and $G_0 = L_0/100$, therefore:

$$k_1 = \frac{\ln(\frac{L_0/2}{L_0/100})}{t_{1/2}} \quad \ldots\ldots (21)$$

OR

$$k_1 = \frac{ln50}{t_{1/2}} \quad \ldots\ldots (22)$$

Similarly, if values of $t_{99/100}$ (the time required for 99% of the gas to be produced) are assumed for both categories of waste, an equation can be derived from which k_2 can be calculated. From equation (18), at $t = t_{99}$ ($L = L_0/100$)

$$k_2 = \frac{\ln(\frac{L_0/2}{L})}{(t - t_{1/2})} \quad \ldots\ldots (23)$$

OR

$$k_2 = \frac{ln50}{(t_{99} - t_{1/2})} \quad \ldots\ldots (19)$$

The values of k_1 and k_2 do not depend on the value chosen for L_o, the amount of gas ultimately produced per mass of refuse. The values are determined by the assumed values of $t_{1/2}$, $t_{99/100}$ and G_o.

The volume of gas produced (for each waste category) prior to any time t can now be calculated from equations (14) and (15), using the calculated values of k_1 and k_2 and the assumed values for $t_{1/2}$ and L_o. The values of L_o for each category are determined as being the product of the category's fractional weight of the total organic content of the waste (on a wet basis) and the assumed total gas production. The derivatives of equations (14) and (15) will yield production rate with time. Total gas production is then simply the sum of gas produced from the individual refuse categories.

The gas generation profile (Figure 3) is quite different from that of the MH and ERL Models in that the profile has two branches; an exponential increase in gas production peaking at time $t=t_{1/2}$, and then an exponential decrease in production. Note that the model does not incorporate a lag-time and production starts at $t_{1/100}$.

3.4 Sheldon-Arleta Model (SA Model)[3]

This theoretical model is very similar to the two-stage, first-

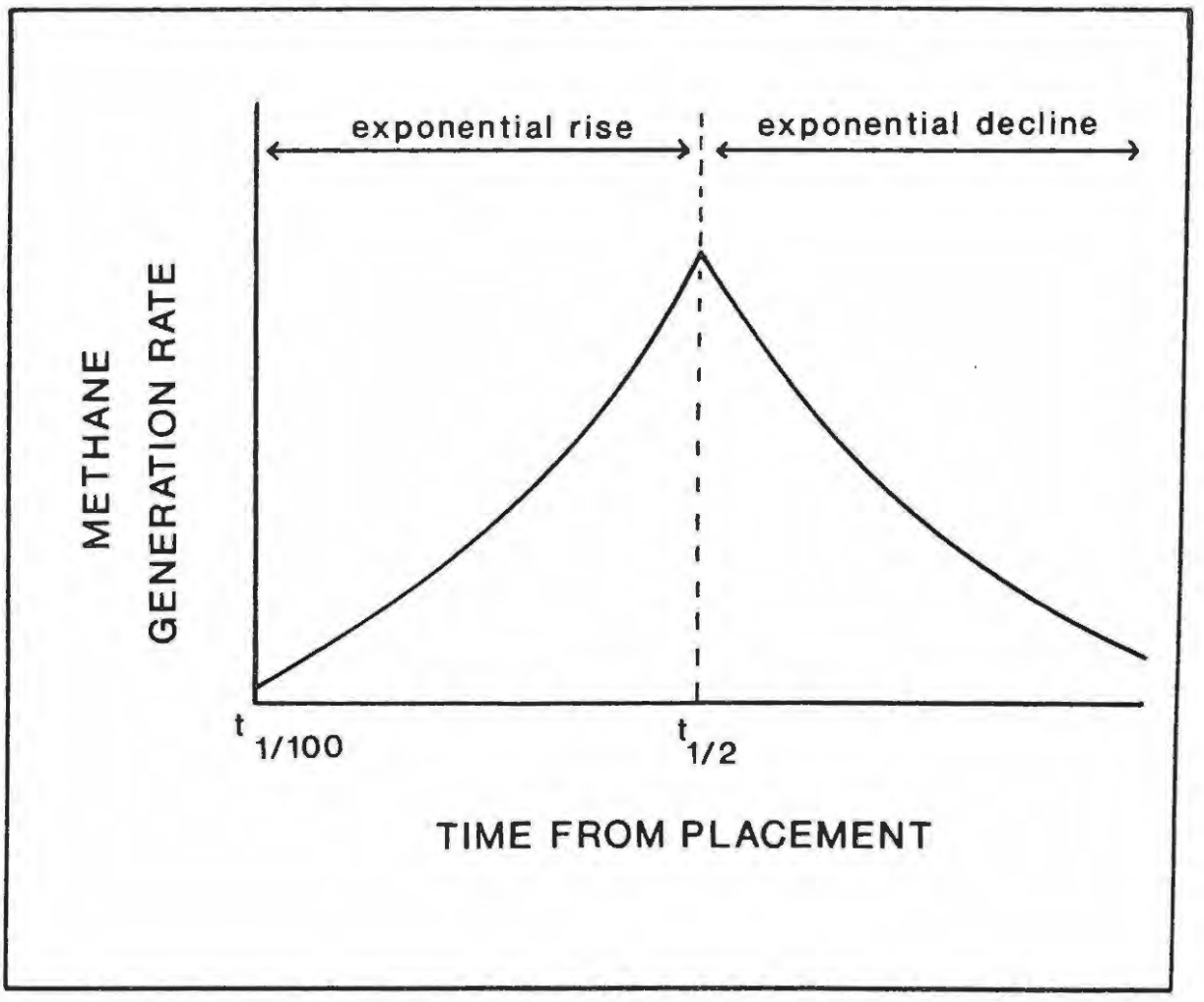

Figure 3. Gas Generation Profile as Predicted by Palos Verdes Model[3]

order kinetic model employed at Palos Verdes. Based on information regarding the anaerobic digestion of sewage sludge, it assumes that the maximum rate of LFG production occurs at 40% of the time required for 99% gasification, and that this time corresponds to the half-life (i.e. $t_{1/2} = 0.4t_{99/100}$). The approach further estimates that 50% of the ultimate yield is attained by the time the gas production rate reaches a maximum, as does the PV Model.

The model is implemented by assuming half-lives for two refuse categories (which consist of 24 fractions), from which the total time required for decomposition can be estimated (Table 2).

TABLE 2
Classification of Biodegradable Refuse in SA Model

DEGRADABILITY	$t_{1/2}$ /yr	t_{total} /yr
readily	9	26
slowly	36	103

The ratio of production rate to the maximum production rate at any time t can then be calculated. A rate versus time curve gives total gas production, and from integration of this curve between the limits t_o and t_{99}, average annual production rates are obtained.

The gas generation profile is identical to that of the PV Model, apart from the fact that gas generation starts from zero and that the half-life is now specified to be $0.4t_{99/100}$ (Figure 4).

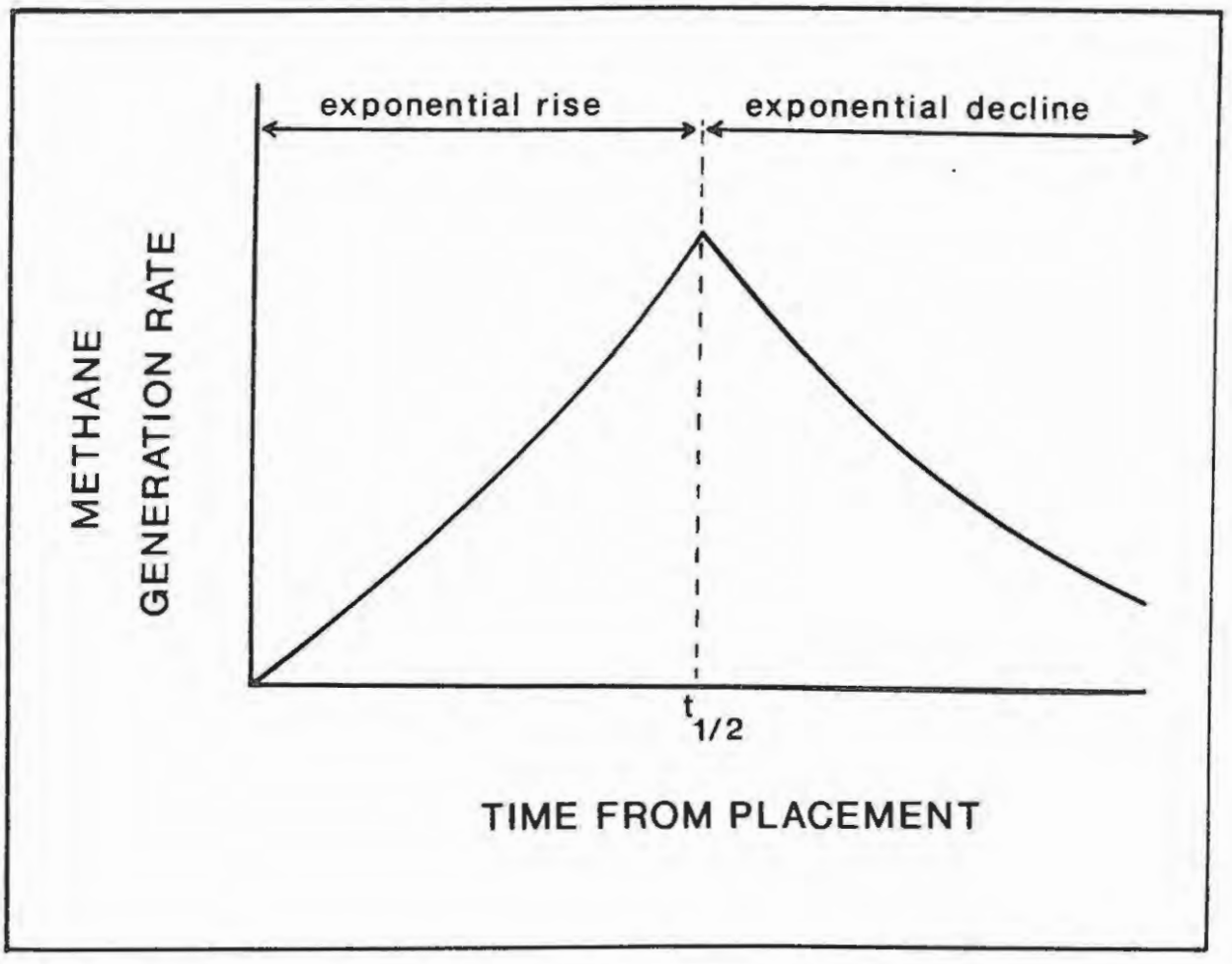

Figure 4. Gas Generation Profile as Predicted by Sheldon-Arleta Model[3]

The model, like the MH and ERL Models, assumes that LFG production is not limited by moisture and nutrients. While this may be true for anaerobic sewage sludge digestion, it most certainly is not the case for landfills.

3.5 Scholl Canyon Model (SC Model)[3]

This model very closely resembles that of the MH Model in that it is a single stage, first-order model based on the kinetics of substrate utilisation. The model was developed before the original Hoeks Model. The rate of methane production is assumed to be at a maximum very shortly after the waste is landfilled, and no account is taken of a lag-time. Just as in the MH Model, gas production rate then decreases exponentially, according to the equation describing substrate limited microbial growth (equation (1)). The gas generation curve is therefore identical in both of these, entirely theoretical, models (Figure 5).

The implementation of the model is slightly different from the approach used in the MH Model. In the MH Model, refuse is quantified in terms of the amount of biodegradable waste in three different categories ($P_{o(j)}$), and the maximum amount of methane from a biodegradable tonne of refuse (α = 416 $m^3.tonne^{-1}$) is then used to quantify the methane potentially available from the refuse. In the Scholl Canyon Model, there is no sub-division of waste into categories describing susceptibility to degradation. The total waste (biodegradable and non biodegradable) is assumed to produce between 60 and 100 m^3 of methane per tonne of refuse.

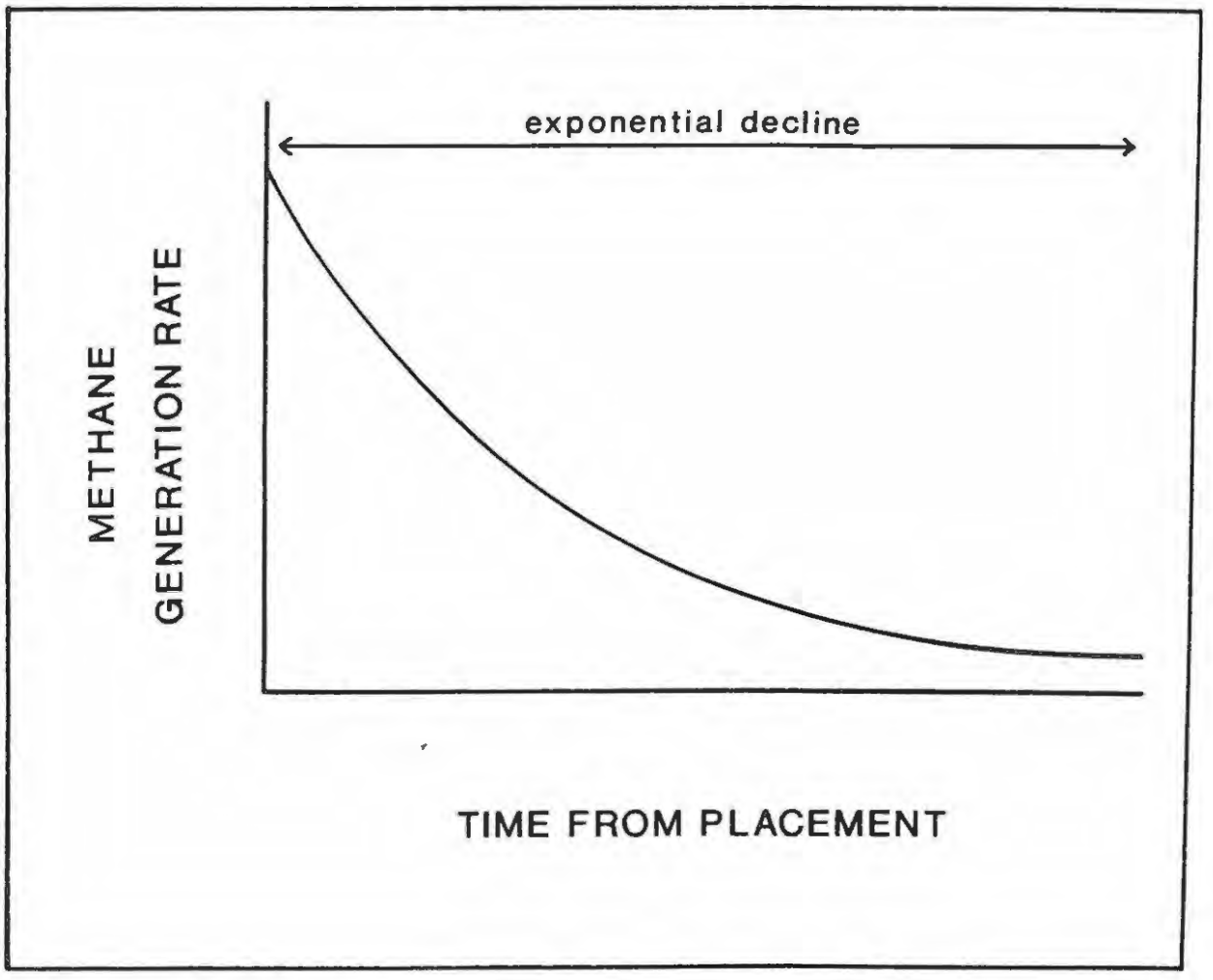

Figure 5. Gas Generation Profile as Predicted by Scholl Canyon Model[3]

This is between 15% and 30% of the value used by the MH Model and therefore appears to account for the biodegradability of refuse. However, no account is taken of the fact that different wastes degrade at different rates and a composite rate constant, k, is thus used. In this respect, the model is more primitive than the MH Model (understandably so, the model being developed approximately 5 years before the original Hoeks Model).

3.6 Gaz de France Model (GDF Model)[1]

This model is based on biochemical laboratory tests and while it can only be rigorously applied to the site for which it was developed (Montaubert, France) the method of approach can be applied to other sites.

The biochemical tests are aimed at determining the specific fermentation kinetics of the site under consideration. Fermentability tests provide data as to:

(i) the quantity of readily decomposable organic matter still remaining

(ii) the quantity of organic matter available for methanogenic bacteria

(iii) the maximum volume of methane which can be obtained by decomposition of the organic matter available to the methanogenic bacteria.

The tests are carried out at a constant temperature of 37-40 °C and conducted for 90 days, when biogas production is observed to

become negligible. Tests also provide data on organic matter composition and moisture contents.

Using the data from the laboratory tests, a kinetic law for the site concerned was established. While the report mentions that the rate of gas production is proportional to the amount of organic substrate available, it never discloses what this kinetic law actually is or what parameters control it. It does, however, give the results of modelling for the Montaubert site, after the model has been adjusted using the current flow rate of biogas from the site. The author has used this data to generate a gas generation profile. The profile of the curve is unlike that yielded by any of the other models (Figure 6). Instead of an exponential increase and decrease in gas production, there is a logarithmic increase, followed by a period of constant gas production and then a logarithmic decrease in generation rate. This indicates that the model is not a first-order model.

3.7 Biokinetic Model (B Model)[2,5]

This model was developed in response to a need to describe and explain data on gas production obtained from the Mountain View Controlled Landfill Project. It became apparent that a simple first-order model was not adequate to describe gas production during the first few years of refuse decay, although it sufficed to describe production in later years.

The model is initially generated theoretically, by simulating the

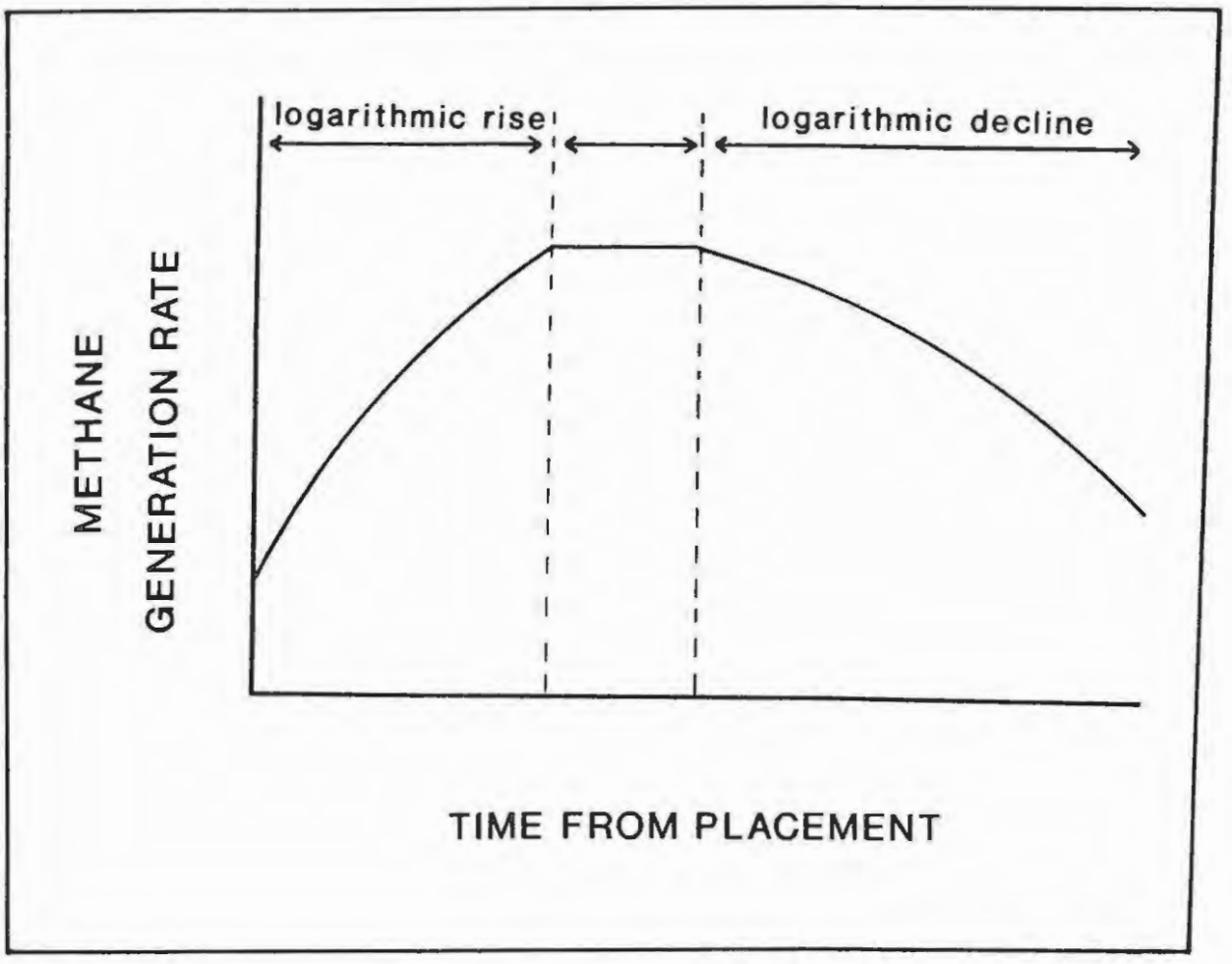

Figure 6. Gas Generation Profile as Predicted by Gaz de France Model[1]

key chemical and biochemical reactions in the refuse ecosystem that lead to methanogenesis. By approximating the solution of these chemical and biochemical equations, a simple generation profile is obtained, consisting of a lag time, a rising hyperbolic branch and a decreasing exponential branch (Figure 7). The rate equations are derived from this profile. (In all the other models discussed, the generation profile is obtained from the rate equations.) The hyperbolic curve, which represents the initial increasing methane generation rate can be described by the function:

$$G_{k(t)} = \coth\alpha_k(t_{2k}-t) - \coth\alpha_k(t_{2k}-t_{0k}) \quad \ldots\ldots(25)$$

$$\text{for } t_{0k} < t \leq t_{1k}$$

and the exponential curve describing decreasing methane production as:

$$G_{k(t)} = G_{pk}\exp(-\lambda_k(t-t_{1k})) \quad \ldots\ldots(26)$$

$$\text{for } t > t_{1k}$$

where:

- t_{0k} = time when gas generation starts
- t_{1k} = time of peak gas generation
- t_{2k} = time at which hyperbolic curve approaches infinity asymptotically
- k = the k^{th} component of refuse
- G_k = generation rate
- G_{pk} = peak generation rate
- λ_k = the time, t_{3k}, required to recover a given fraction of the total generation potential (a function of the rate constant)
- α_k = factor determining the shape of the rising branch of the generation function; high values produce an almost linear rise (dependant on the nature of the waste)

Gas generation rates can be calculated by applying equations (25)

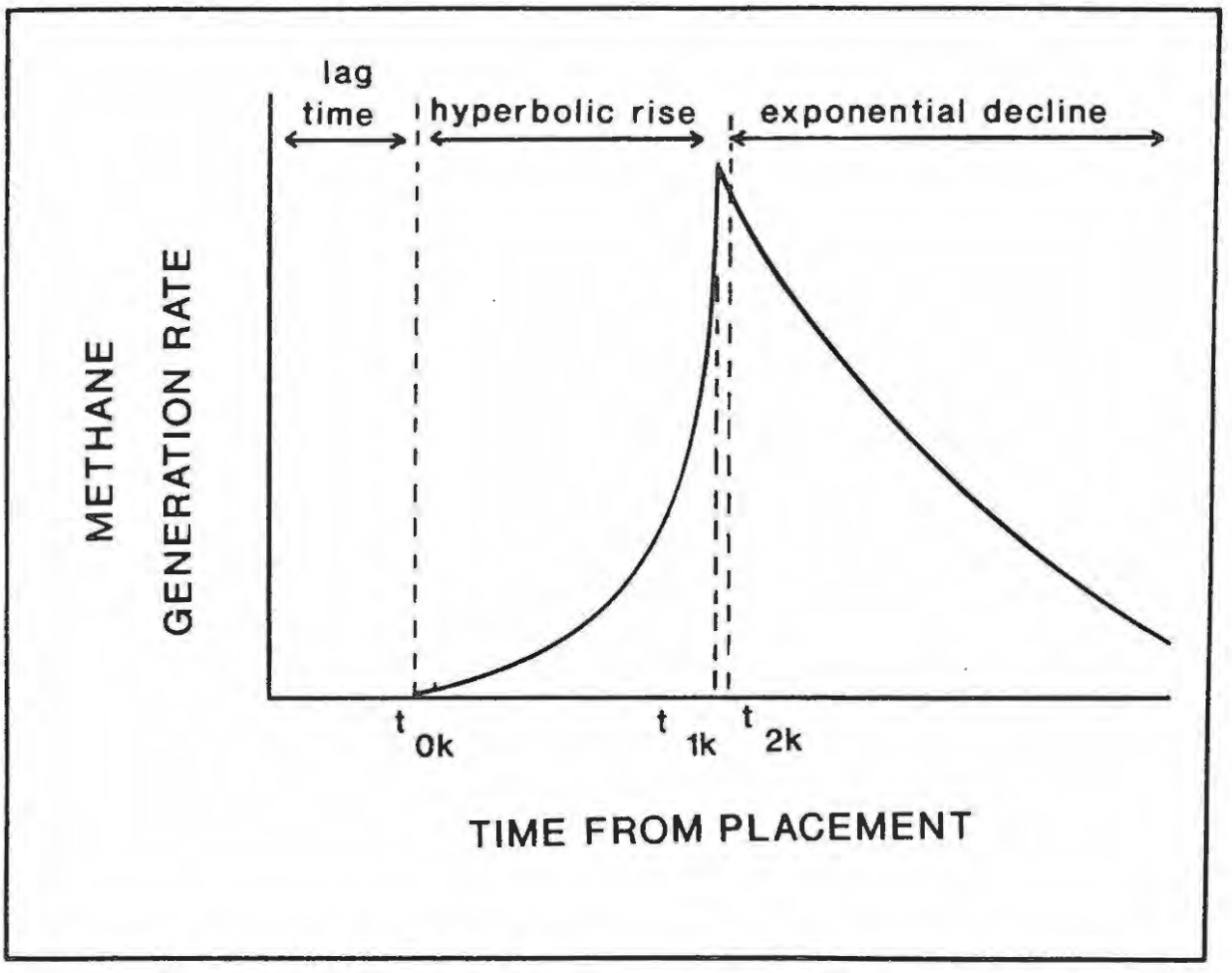

Figure 7. Gas Generation Profile as Predicted by Biokinetic Model[2,5]

and (26), once values for α_k are selected for different grades of refuse, and λ_k, t_{2k}, t_{ok} and t_{1k} are obtained experimentally or by calculation.

Once the simulated gas generation potential for the Mountain View Landfill Site had been modified to account for the non-hydrolysable fraction of refuse, the biodegradability factor and the water content of the refuse, it correlated fairly well with past and present field measurements of flow rates. These field measurements had, as yet, only been conducted for a short period of 3 years.

Researchers who developed this model are aware that it is a model essentially based on field data, making the model very site specific and of limited general applicability. The empirical function does not explicitly include rate-determining variables such as moisture, temperature etc.

A modified version of the model has therefore been developed to include the effects of these variables. This was done by incorporating chemical/biokinetic feedback loops, for the chemical parameter influence on microbiological rate processes. This integrated biokinetic/gas transport model is based on the principles governing the biokinetics of the landfill environment and on the physics of gas migration. The predictions of gas generation using this modified model correlate slightly better to the Mountain View Landfill data than those of the earlier model.

3.8 EMCON MGM Model (E Model)[9]

The development of this model is not documented in any of the literature reviewed. The model is included for reasons of comparison with the B Model and because it has some field verification of its predictions from a minimum of three different sites.

The gas generation profile includes three stages: a lag-time, a short period of linearly increasing gas production and a gradual exponential decline (Figure 8). It is thus very similar to the B Model, the only difference being that in the B Model, the period of increasing gas production is described as hyperbolic. However, the B Model does state that the higher the value of the parameter α_k (equation (25)), the more linear the rise of this hyperbolic function becomes. So for high values of αk, the models are essentially the same.

The fact that predictions made with this model closely resemble field measurements taken at the Mountain View Landfill Site, as do those of the B Model, indicates that both these models closely approach reality.

4. COMMENTS ON MODELS

The comments which follow, for each individual model, are given in point form, as a type of summary.

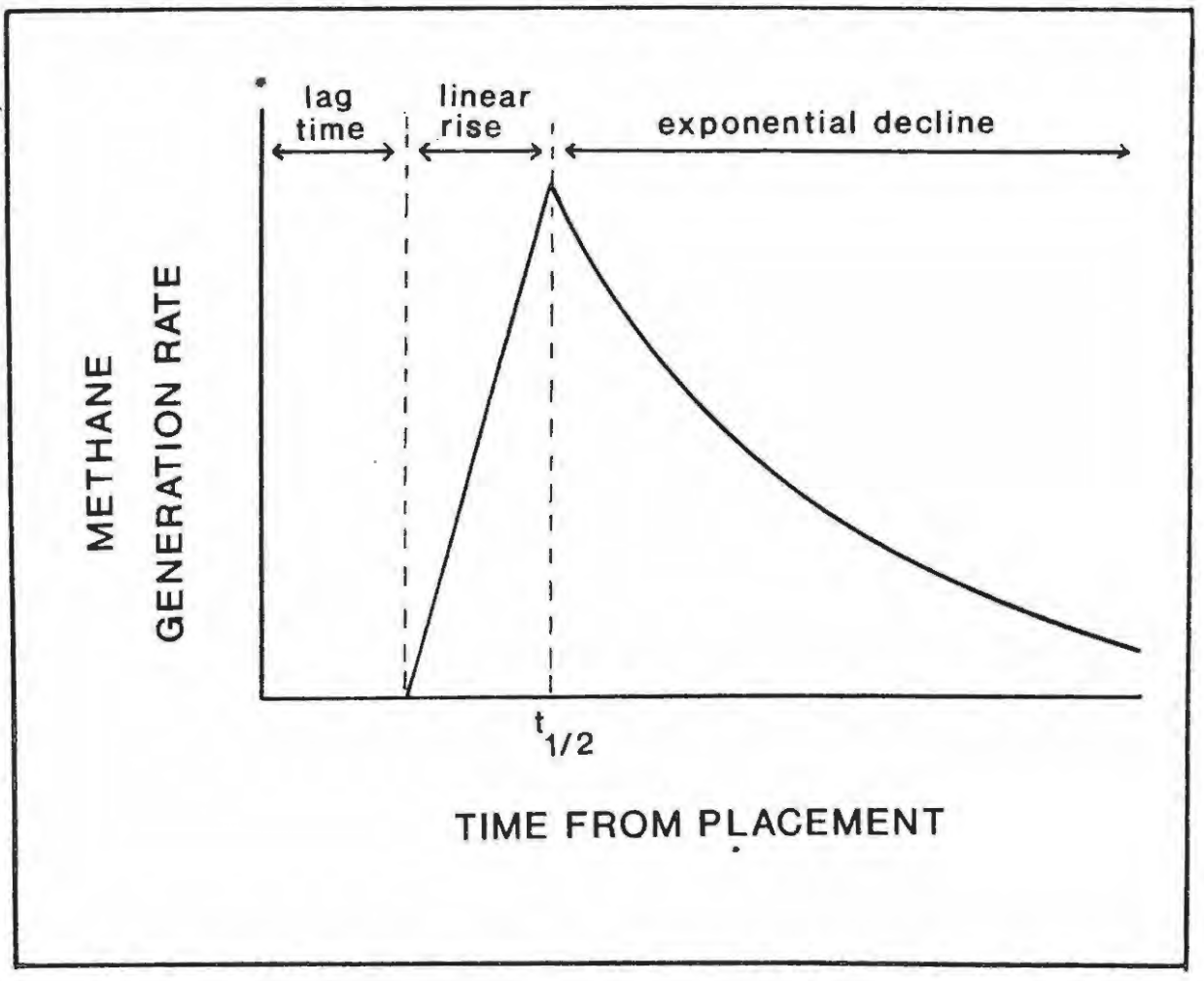

Figure 8. Gas Generation Profile as Predicted by EMCON MGM Model[9]

Modified Hoeks Model

(i) One stage of gas generation, with rate starting at a maximum and decreasing exponentially .

(ii) $t_{1/2}$ values assumed for three categories, based on degradation susceptibility.

(iii) Maximum gas production assumed (α = 416 m^3 CH_4.$tonne^{-1}$), hence the assumption of ideal conditions, with no consideration of rate-controlling factors.

(iv) Entirely theoretical, no field verification.

Environmental Resources Limited Model

(i) A stage of aerobic decay considered, followed by methane production starting at a maximum and decreasing exponentially.

(ii) $t_{1/2}$ values assumed as for MH Model for the anaerobic reaction. For the aerobic reaction a half-life of 8 weeks is assigned and the reaction assumed to continue for 5 half-lives (40 weeks).

(iii) Rate equation is for maximum methane production under ideal conditions, with no consideration of rate-controlling environmental factors.

(iv) Implementation not entirely theoretical as site hydrogeology and pumping trials constrain predictions.

Palos Verdes Model

(i) Two stages of gas generation, with production rate increasing exponentially, once 1% of the ultimate gas yield has been produced, to a maximum at time $t = t_{1/2}$, and then decreasing exponentially.

(ii) $t_{1/2}$ values assumed for three categories; very low values (estimate that gas is produced from a landfill site for only 6 years).

(iii) Ultimate gas yield assumed, on wet refuse basis, with no consideration of rate-controlling factors.

(iv) Theoretical prediction, no field verification.

Sheldon-Arleta Model

(i) Two stages of gas generation, with production rate increasing exponentially to a maximum at time $t = t_{1/2} = 0.4t_{99/100}$, and then decreasing exponentially.

(ii) $t_{1/2}$ values assumed, based on observations; values very high.

(iii) No account taken of moisture and nutrient limitations as model is based on anaerobic sewage sludge digestion.

(iv) Theoretical prediction, no field verification.

Scholl Canyon Model

(i) Gas generation profile shows one stage, with generation rate starting at a maximum and decreasing exponentially.

(ii) $t_{1/2}$ fitted to observed data by trial and error and composite k value hence obtained; no refuse categorisation according to degradation susceptibility.

(iii) No account taken of rate-controlling environmental factors.

(iv) Prediction largely theoretical, but not entirely so, as $t_{1/2}$ values are based on field observations.

Gaz de France Model

(i) Generation profile has three stages: logarithmic increase to maximum generation rate, constant rate at maximum and logarithmic decrease.

(ii) $t_{1/2}$ values obtained from laboratory fermentation experiments.

(iii) No account of temperature and moisture limitations, these being controlled in the laboratory experiments.

(iv) Predictions calibrated to current flow rate data collected in the field; no actual field verification.

Biokinetic Model

(i) Simulated gas generation profile, based on chemical and biochemical reactions, includes a lag-time, followed by increasing production to a maximum, according to a hyperbolic function, followed by a gradual exponential decrease.

(ii) The rate coefficient is determined by calculation, based on the shape of the generation profile.

(iii) The model includes chemical/biokinetic feedback loops to account for the influence of rate-controlling variables.

(iv) Predictions verified with short-term field data.

EMCON MGM Model

(i) Generation profile shows two stages of gas production after initial lag-time: linearly increasing gas production rate to a maximum, and then gradual exponential decay.

No further comments can be made on this model as the author is not aware of how the model was developed. The predictions have been validated with short-term field data from a number of sites.

5. CONCLUSIONS

Discussion of the models has focused on what the shape of the gas generation profile is. This profile merely indicates the pattern of gas production for one particular batch of waste, deposited at a particular time, as it undergoes degradation. Obviously a landfill contains countless numbers of these batches (or cells) so that countless numbers of generation profiles can be described. The individual profiles are of little importance in predicting gas generation from a landfill. What is important, is the cumulative effect of these profiles, rather than the profiles themselves.

While batch generation profiles may differ significantly between models, the cumulative effect of these profiles does not vary

significantly between models, especially over the longer term[9]. The cumulative profiles (for the case where refuse is landfilled at a constant rate over time) generally take the form of a logarithmic increase in production rate, which approaches a constant rate at between 30 and 50 years, and then decreases exponentially when landfill operations cease (Figure 9).

There is still much room for improvement in the modelling of landfill methane generation. Better correlation between rate controlling variables, such as moisture and temperature, and methane generation are necessary. The Biokinetic Model (section 3.7) appears to have attempted this with a measured degree of success, although field data at this stage is still somewhat limited and can therefore not ensure a complete verification of the model. The need for long-term gas generation data, from a variety of different sites (wet, dry, hot, cold etc.), is a very real one and must be addressed before models can be satisfactorily implemented. Once this data is obtained, site-specific models could be defined for a number of different site types. Sites could be classified as follows:

(i) wet and hot
(ii) dry and hot
(iii) wet and cold
(iv) dry and cold

This would be an initial attempt to develop a universal set of models.

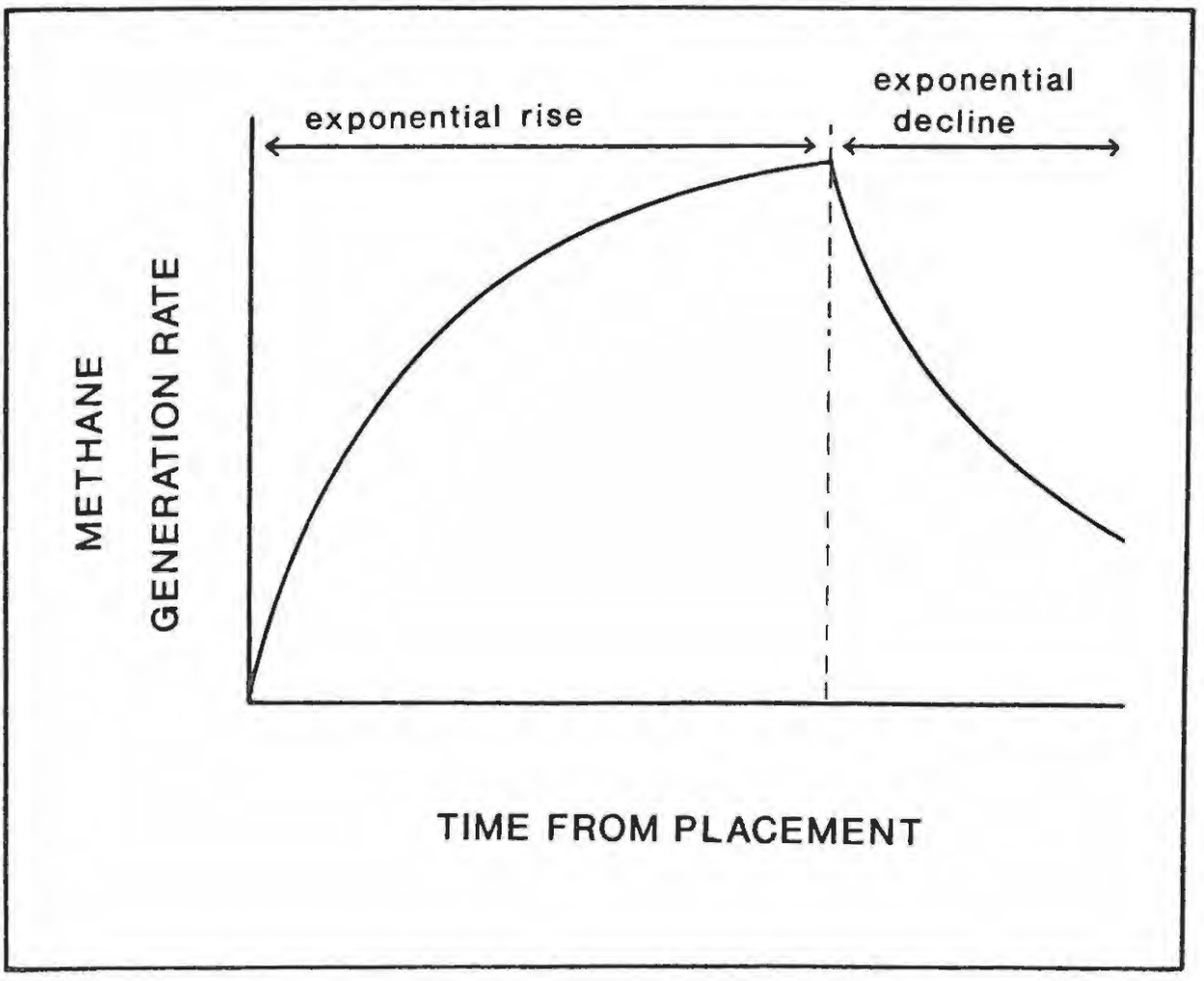

Figure 9. Cumulative Gas Generation

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CHAPTER 10

ASSESSMENT OF METHANE POTENTIAL

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1. INTRODUCTION

The need to assess the present and future methane potential of a landfill site before embarking on a landfill gas extraction scheme has already been mentioned in chapters 8 and 9. The accurate prediction of methane generation rates over time enables decisions to be made as to the extraction rates that can be employed and the length of time for which gas can be extracted. The feasibility of a proposed extraction scheme can hence be assessed.

The rate at which gas is pumped out of the landfilled refuse must not exceed the normal rate of LFG production. If pumping rates in excess of the generation rate are employed, air will be drawn into the refuse, replacing the extracted gas. This can lead to the subsequent poisoning of anaerobic methanogens and a decrease in methane generation.

In this work, two approaches are used to assess the possible methane generation rates at the Grahamstown Landfill Site; a theoretical one in which generation is mathematically modelled according to kinetic equations, and an empirical one in which gas pumping experiments are performed*. Ideally, the predictions from the theoretical approach should support the results from the pumping trials.

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*T.M. Letcher and the author used a similar approach to assess the LFG potential of the Kya Sand Landfill in Randburg in August 1991 (see attached report at the end of this thesis).

2. A THEORETICAL ASSESSMENT

Some of the kinetic models which have been developed for the prediction of methane generation rates have been discussed at length in Chapter 9.

2.1 Methane Potential of the Grahamstown Landfill Site

In order to assess the energy available from the Grahamstown Landfill, it was decided to use the Modified Hoeks Model (MH Model), as this is a simple model with minimal computing demands. It is, unfortunately, an entirely theoretical model, which takes no account of rate controlling factors such as moisture (hence its simplicity).

2.1.1 Ideal Predictions based on MH Model

The results of an extensive refuse survey carried out at the Grahamstown Landfill (Chapter 8) form the basis of the kinetic analysis of methane production at the site. Amounts of dry biodegradable refuse ($P_{o(j)}$) and relative half-lives ($t_{1/2(j)}$), as given by Hoeks[4], are detailed in Table 1. Using these values and assuming that tipping takes place at the same rate each year, the total rate of methane production from the three types of refuse at any time t after the first refuse was deposited, is given by the following equation:**

**This equation is identical to equation (9) in Chapter 9.

$$\frac{d[CH_4]}{dt} = Q_t = \sum_{t=1}^{t=t} (416x0.693x910)\exp(-0.693xt)$$

$$+$$

$$\sum_{t=1}^{t=t} (416x0.693x970/5)\exp(-0.693xt/5)$$

$$+$$

$$\sum_{t=1}^{t=t} (416x0.693x3080/15)\exp(-0.693xt/15) \ldots\ldots (1)$$

TABLE 1
Analysis of Biodegradable Refuse

(j)	WASTE TYPE	$t_{1/2(j)}$ yr	AMOUNT tonne.yr^{-1}	MOISTURE %	$P_{o(j)}$ / dry tonne.yr^{-1}
(1)	food	1	2 470	63	910
(2)	garden	5	1 930	50	970
(3)	paper+wood	15	4 000	23	3 080

The resultant methane production at the Grahamstown Landfill Site, if tipping is continued at this rate for 50 years, is obtained by adding each year's contribution, and is graphed in Figure 1. The model indicates that after 5 years of landfilling (from 1986 to September 1991), methane will be produced at a rate of approximately 80 m^3.hr^{-1}***. If tipping were to continue indefinitely at the same rate, a steady state of methane production, 240 m^3.hr^{-1}, would eventually be realised. In the event of tipping being discontinued after the 50th year, the model indicates that methane will still be produced in the 100th

***This is equivalent to 30 m^3 methane per dry biodegradable tonne per year.

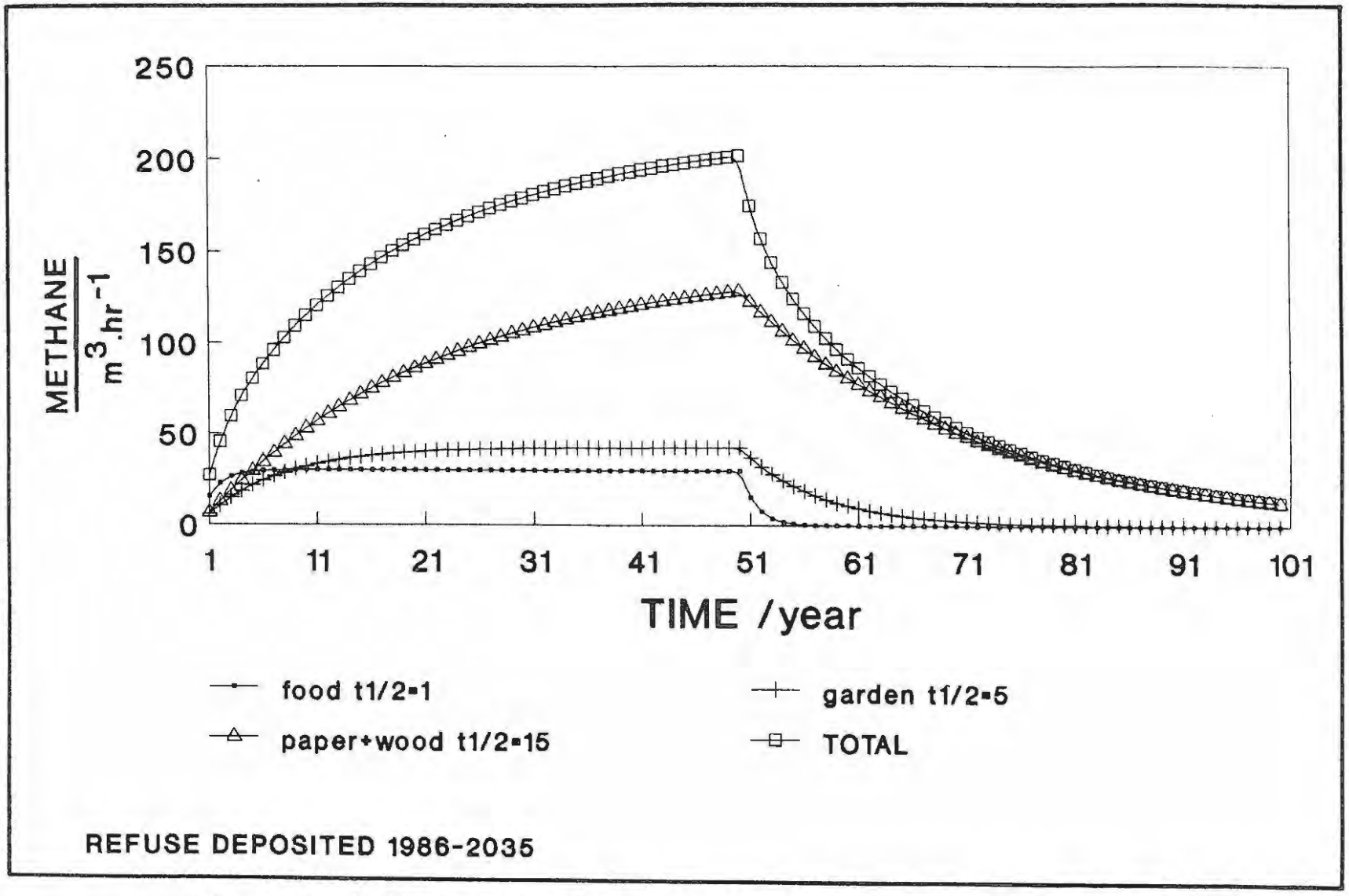

Figure 1. Methane Generation from the Grahamstown Landfill as Predicted by the MH Model.

year, at the rate of only 10 $m^3.hr^{-1}$.

It must be stressed that these are maximum predictions for methane generation, as α has been given the value of 416 m^3 methane per tonne of dry biodegradable waste. The predictions will only hold if the values used for $P_{o(j)}$, $t_{1/2(j)}$ and α are in fact accurate.

2.1.2 Range of Predictions based on MH Model

In an attempt to determine model sensitivity to the parameters $P_{o(j)}$, $t_{1/2(j)}$ and α, and to determine a realistic range of predictions based on the MH model, minimum and maximum values for the parameters $P_{o(j)}$, $t_{1/2}$ and α have been assigned as follows:

(i) The theoretical maximum value for the parameter α is 416 m^3 CH_4 per tonne dry biodegradable waste. A minimum value for α (possibly a more realistic value) of 100 m^3 per tonne dry biodegradable waste is used here. (Literature values obtained from lysimeter experiments range between 50 and 250 $m^3.tonne^{-1}$[3,7].)

(ii) The $P_{o(j)}$ values used in equation (1) are those determined from the refuse survey. In the following calculations,it is assumed that the $P_{o(j)}$ values are only known to within ± 20% of the experimental value.

(iii) The $t_{1/2}$ values in equation (1) are those used by Hoeks[4]. Assuming that the possible maximum and minimum values vary by a factor of four, the maximum has been obtained by

multiplying the Hoeks[4] values by two, and the minimum has been obtained by dividing it by two.

The introduction of a range of possible values (Table 2) for these three parameters, allows for the determination of eight composite generation curves.

TABLE 2
Upper and Lower Limits for α, $t_{1/2}$ and $P_{o(j)}$

α min $m^3.tonne^{-1}$	α max $m^3.tonne^{-1}$	(j)	$t_{1/2(j)}$ min yr	$t_{1/2(j)}$ max yr	$P_{o(j)}$ min tonne. yr^{-1}	$P_{o(j)}$ max tonne. yr^{-1}
100	416	(1)	0.5	2	728	1 092
		(2)	2	10	776	1 164
		(3)	7	30	2 464	3 696

The four graphs in Figure 2 representing the two upper and the two lower limits were generated by using the data in columns 2, 4, 5 and 7, and data in columns 1, 4, 5 and 6 respectively. The reason for using both a minimum and maximum value for $t_{1/2}$, for each limit, is because the minimum $t_{1/2}$ value produces a maximum generation rate during the tipping period, while the maximum $t_{1/2}$ value produces a maximum generation rate once tipping is terminated (see Figure 2). (The lower the value of $t_{1/2}$, the more rapid the increase to a maximum generation rate, and the quicker the decline once tipping is terminated. The higher the value of $t_{1/2}$, the slower the increase to a maximum generation rate and a correspondingly slower decline in gas production.)

Under the conditions allowed for, the maximum possible rate of

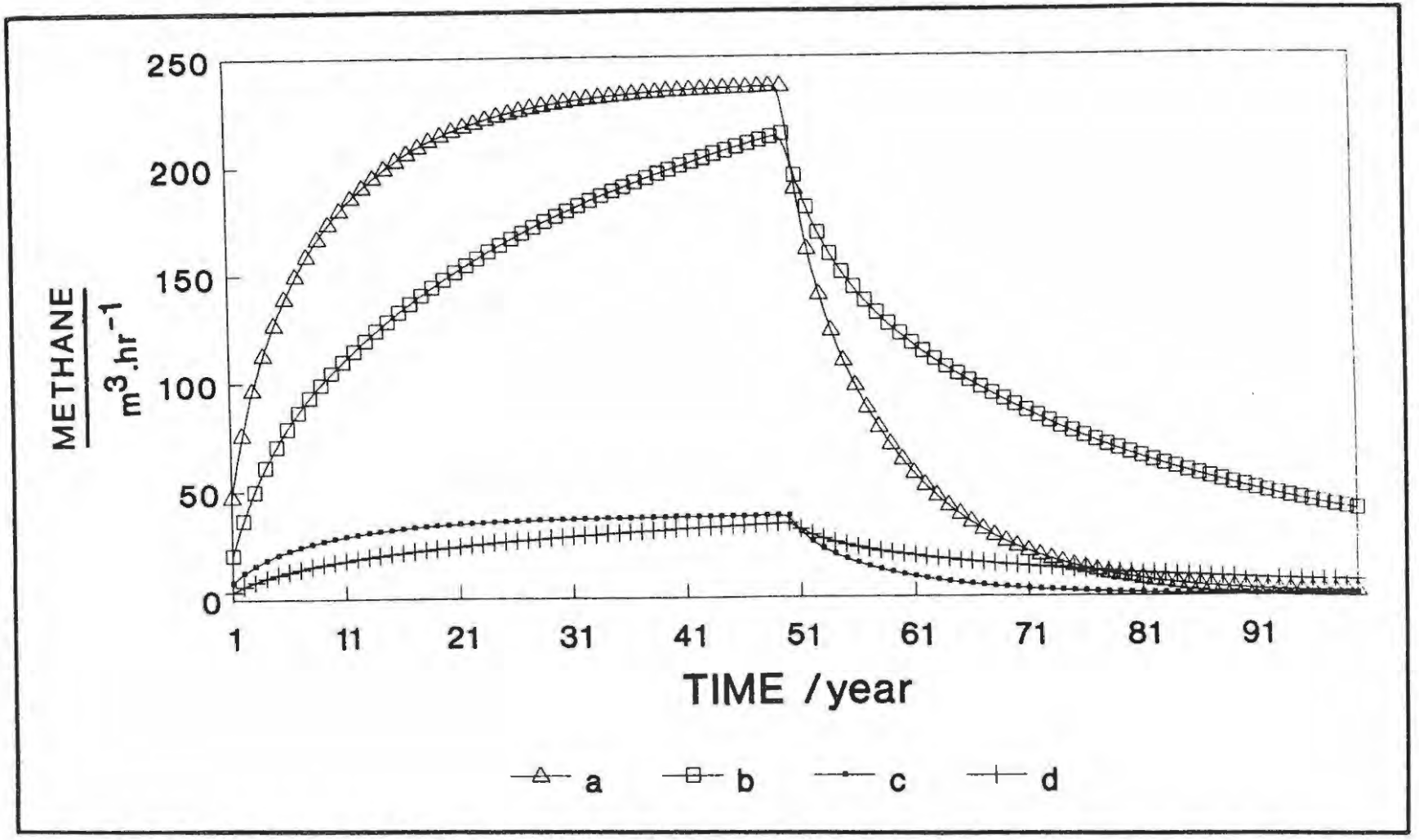

Figure 2. Upper and Lower Limits of Methane Generation from the Grahamstown Landfill

a - $t_{1/2(j)}$ minimum, $P_{o(j)}$ maximum, α maximum
b - $t_{1/2(j)}$ maximum, $P_{o(j)}$ maximum, α maximum
c - $t_{1/2(j)}$ minimum, $P_{o(j)}$ minimum, α minimum
d - $t_{1/2(j)}$ maximum, $P_{o(j)}$ minimum, α minimum

methane generation, before tipping is discontinued (year 50), is 230 $m^3.hr^{-1}$, while in the 100th year the maximum possible rate is 40 $m^3.hr^{-1}$. The minimum limits are 40 $m^3.hr^{-1}$ and 0 $m^3.hr^{-1}$ respectively. The large range of the maximum prediction, 40-230 $m^3.hr^{-1}$, which is equivalent to a deviation of between +15% and -80% from the ideal prediction (200 $m^3.hr^{-1}$), emphasises the model sensitivity to values of α, $P_{o(j)}$ and $t_{1/2(j)}$. The results show that the model is far more sensitive to changes in α and $P_{o(j)}$ than it is to changes in $t_{1/2}$ values, hence the need accurately to quantify $P_{o(j)}$ and α. While the former can be done by executing extensive refuse surveys and laboratory tests, the latter is a function of an intricate network of site-specific factors and is therefore difficult or impossible to determine experimentally.

2.1.3 Predictions based on ERL Model[5]

The MH Model, which has been used to predict methane generation rates, does not include a lag-time after waste placement, during which aerobic decomposition of the waste takes place before methanogenesis sets in. A model which does incorporate the concept of a lag-time is the Environmental Resources Limited Model (ERL Model[5]). The data in Table 1 were used to generate a predictive curve based on the rate equation which forms the basis of the ERL Model:

$$\frac{d[CH_4]}{dt} = \sum_{j=1}^{j=3} \sum_{t=1}^{t=t} \frac{416x0.693}{t_{1/2(j)}} xP_{0(j)} \exp\left(\frac{-0.693(t-t_a)}{t_{1/2(j)}}\right) x$$

$$\exp\left(\frac{-0.693xt_a}{t_{1/2(a)}}\right) \ldots\ldots\ldots\ldots\ldots\ldots (2)$$

where: t_a is the time when the aerobic reaction is complete and the anaerobic reaction begins.
$t_{1/2(a)}$ is the half life for the aerobic reaction.

The values used for t_a and $t_{1/2(a)}$ are the same as those assumed by the researchers who developed the model, 0.81 years (42 weeks) and 0.15 years (8 weeks) respectively. The outcome of the modelling is graphed in Figure 3.

The effect of considering an aerobic phase of degradation which lasts for 42 weeks after the placement of waste, is quite remarkable. A maximum of only 5.5 m^3 of methane is produced per hour, after 50 years of tipping. This is no more than 3% of the amount predicted by the MH Model, 200 $m^3.hr^{-1}$, and is outside the predictive range (40-230 $m^3.hr^{-1}$). Model sensitivity to the amount of substrate available for the methanogens ($P_{o(j)}$) is thus once again demonstrated. It is understandable that the models should be so sensitive to this parameter, being models based on first-order, substrate limited kinetic equations.

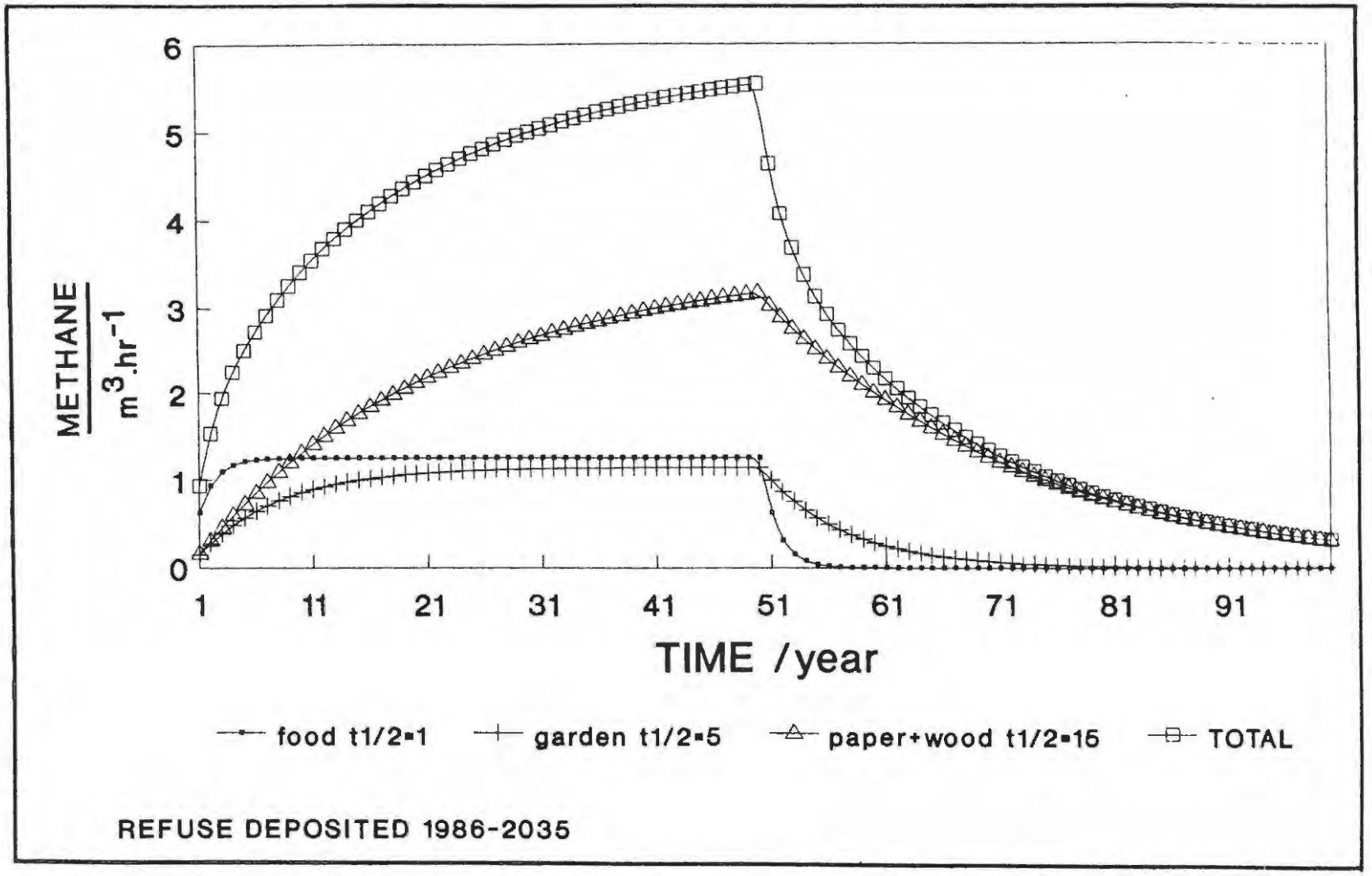

Figure 3. Methane Generation from the Grahamstown Landfill as Predicted by the ERL Model.

2.2 Methane Potential of a Rehabilitated Portion of the Grahamstown Site

Theoretical determination of methane production from a small portion of the Grahamstown Landfill which received refuse for three years (1986-1988), and has since been rehabilitated with grass cover (see Photograph 2 and Figure 3, Chapter 4) has been carried out using the MH Model. The quantities of biodegradable refuse in this site (Table 3) were determined on the basis of the site dimensions (6 500 m^3) and data from the refuse survey. Results of the modelling are graphically illustrated in Figure 4.

TABLE 3
Analysis of Biodegradable Refuse in Rehabilitated Site

(j)	WASTE TYPE	$t_{1/2(j)}$ year	AMOUNT tonne.yr^{-1}	MOISTURE %	$P_{o(j)}$ /dry tonne.yr^{-1}
(1)	food	1	220	63	80
(2)	garden	5	180	50	90
(3)	paper+wood	15	300	23	230

Predictions from the model are that at present (year 6) the ideal rate of methane production is 2.5 $m^3.hr^{-1}$ (using data from Table 3, column 3 and 6, where α = 416 m^3 per dry biodegradable tonne). The maximum and minimum limits are 2.9 and 0.5 $m^3.hr^{-1}$ respectively, according to the prediction range established in section 2.1.1.

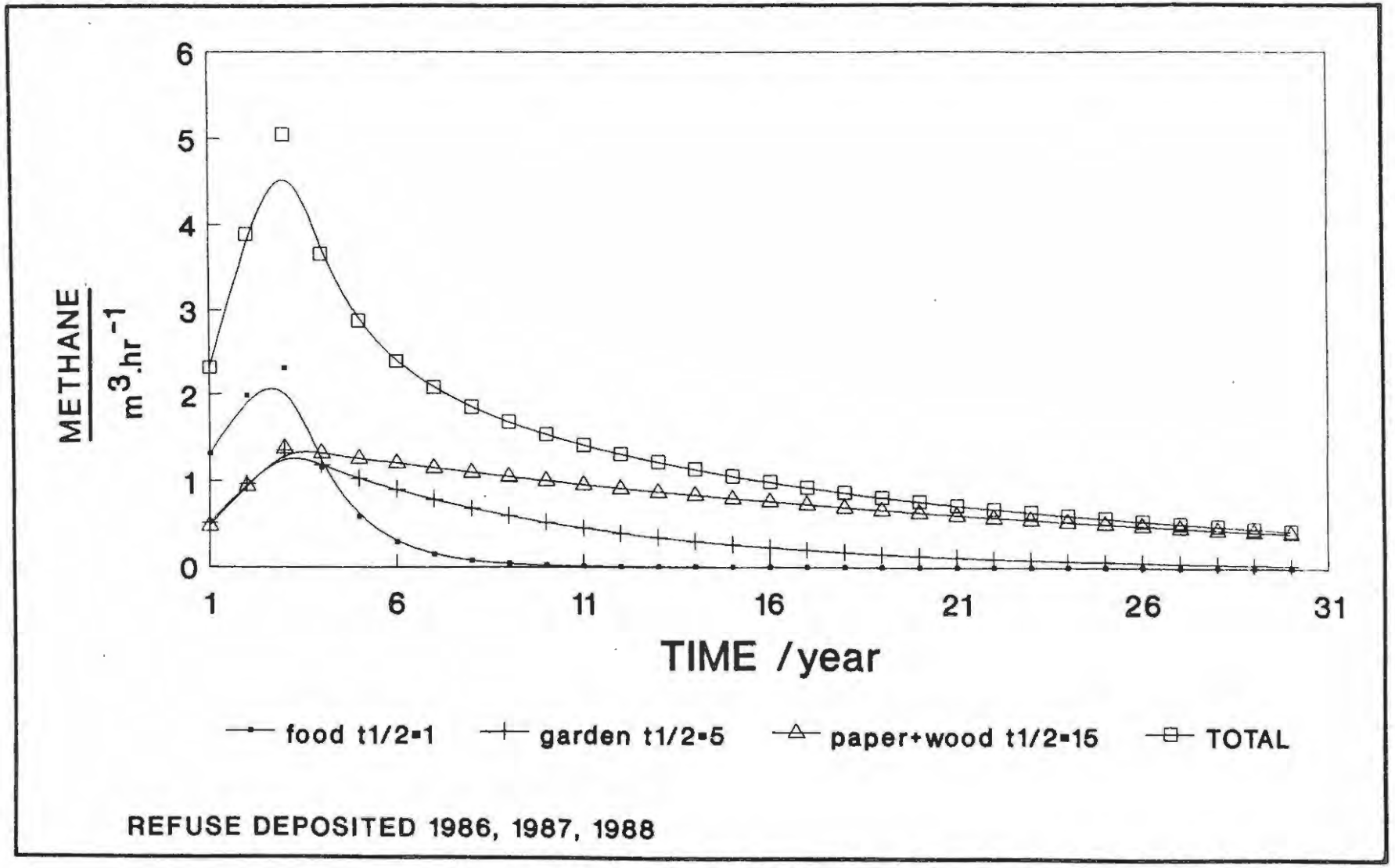

Figure 4. Methane Generation from the Rehabilitated Portion of the Grahamstown Landfill as Predicted by the MH Model.

3. AN EMPIRICAL ASSESSMENT OF THE METHANE POTENTIAL OF A LANDFILL SITE

Pumping experiments coupled with sensitive pressure measurements and sophisticated gas analysis can be used to assess the potential of a landfill site. In this section, (i) the natural flow rate of LFG from gas extraction wells in the site is compared to the rate of production of methane and (ii) the pumping methods used to assess a landfill's methane potential are discussed.

3.1 Measurement of Flow Rates at the Grahamstown Landfill Site

The monitoring of flow rates at the Grahamstown Landfill has been discussed in detail in Chapter 7. The results are discussed here because of their relevance to the prediction of methane generation rates.

The results of the numerous anemometer flow measurements (see Chapter 7) indicate that the average natural flow rate, Q_n, of gas through the wells in the rehabilitated section of the Grahamstown landfill is 2 $m^3.hr^{-1}$. Average methane content is 35% by volume, giving a methane flow of 0.7 $m^3.hr^{-1}$. This is 30% of the ideal generation rate predicted by the model (2.5 $m^3.hr^{-1}$), and within the range of the prediction (2.9 - 0.5 $m^3.hr^{-1}$). The correlation is a reasonably good one if one considers that in the monitoring technique used, only gas escaping from the open gas extraction wells was monitored. Some gas is known to escape

through the landfill surface and sides (Chapter 4). One therefore expects the natural methane flow rate, Q_n, to be significantly lower than Q_o, the rate of production of methane in the landfill (Figure 5a), because of the amount, Q_a, escaping through the sides and the cover (see Figure 5b).

From flow rate measurements conducted in the operational area of the landfill, it has been estimated that the methane flow from this area of the site is presently 44 $m^3.hr^{-1}$ (Chapter 7). This is 57% of the ideal generation rate predicted by modelling (77.5 $m^3.hr^{-1}$), if methane production from the 2 800 tonne of refuse on the edges, roads etc. (area E, Figure 3, Chapter 4) is not included. The correlation between empirical and theoretical assessments is reasonable considering the limitations inherent in the method of monitoring flow rates.

3.2 Pumping Trials

3.2.1 Continuous Pumping Experiments at the Grahamstown Landfill Site

This method merely gives an indication as to the rate at which LFG is generated in a site. It entails continuous extraction of the gas for as long as possible (2-90 days). If a drop in the methane content and an increase in the oxygen content of the extracted LFG is observed while pumping, then it can be concluded that gas is being extracted at a greater rate than it is being generated (provided the site is well covered). If no

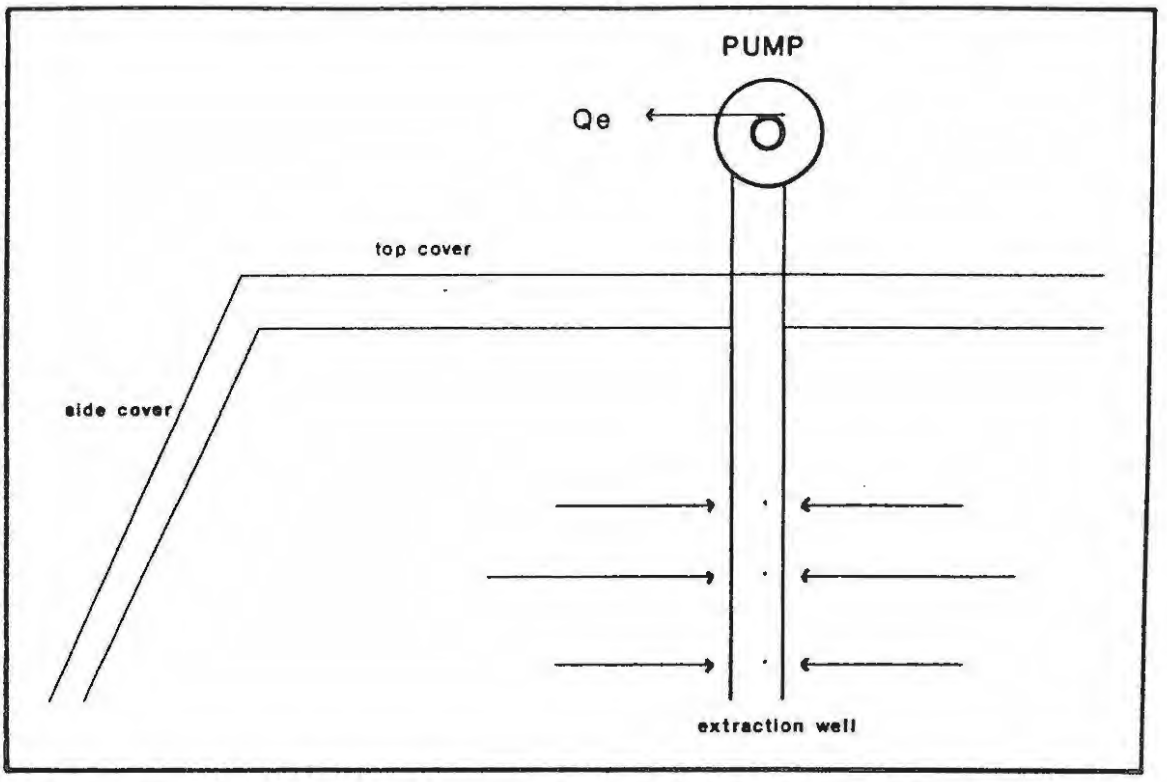

Figure 5(a). Extraction of LFG at the rate at which it is produced, Q_a.

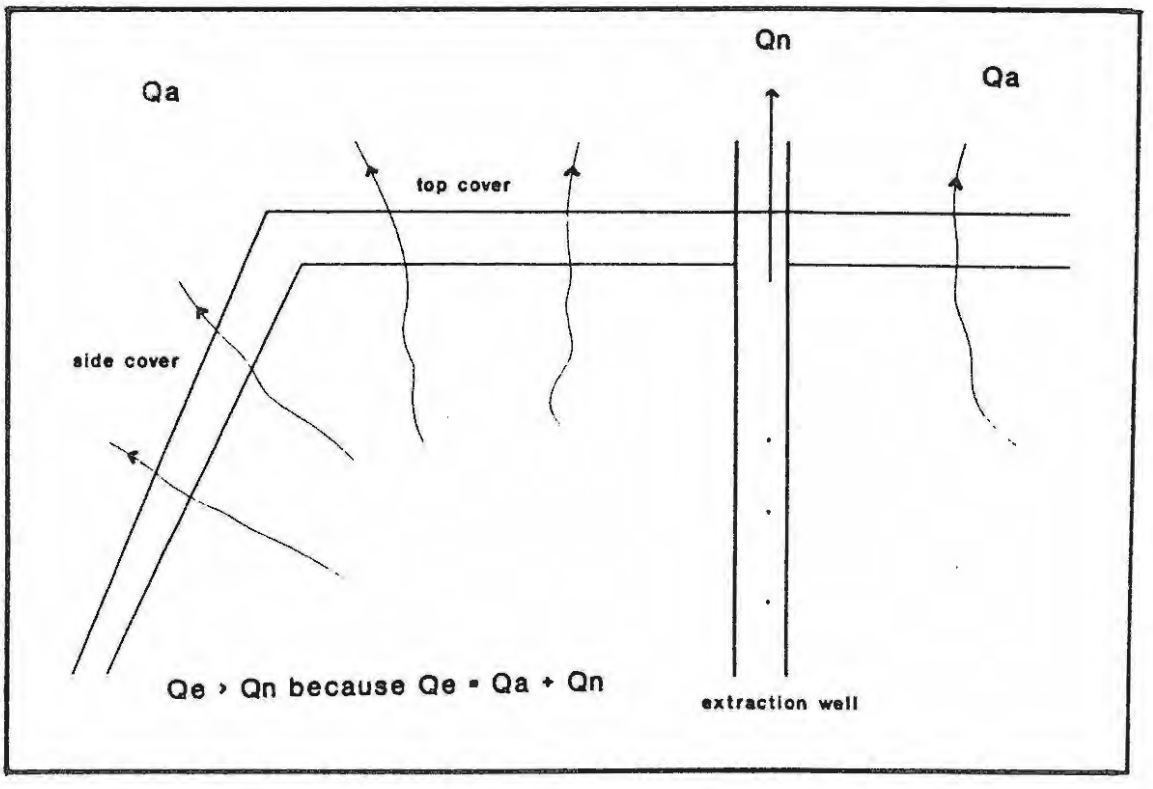

Figure 5(b). The natural flow rate of gas from an open gas extraction well (Q_n) and through the sides and cover of the site (Q_a).

change in the gas composition is observed, then the generation rate is greater than or equal to the extraction rate. Ideally, the extraction pump should be capable of operating at variable speed. If the initial pumping rate is ascertained to be less than the generation rate, then the pumping rate can be sequentially increased, until such time as the methane content just decreases and the oxygen content just increases. This pumping rate, Q_e, is equivalent to the equilibrium pumping rate for the particular well and is equal to the rate of LFG production for the region serviced by the well.

Pumping trials of this nature have been carried out on the rehabilitated portion of the Grahamstown Landfill. Unfortunately, the extraction pump used at the site operates at a fixed vacuum (8 cm water), so that it is not possible to vary the extraction rate (50 $m^3.hr^{-1}$). (A throttle should be included in the gas pipe between the pump and the wells to allow for variation in the pumping rates.) The experiments still proved worthwhile, however, in that the rate of gas generation was ascertained to lie below a certain limit.

The results of one such pumping experiment, during which gas was extracted at 50 $m^3.hr^{-1}$ for 16 hours from well A1, are graphically illustrated in Figure 6. The methane content of the gas was initially very low, about 20% by volume, and no oxygen was detected to be present. After pumping at 50 $m^3.hr^{-1}$ for 15 hours, the methane content had dropped to 10% and oxygen had increased to 7% by volume. The indication is that the rate of LFG

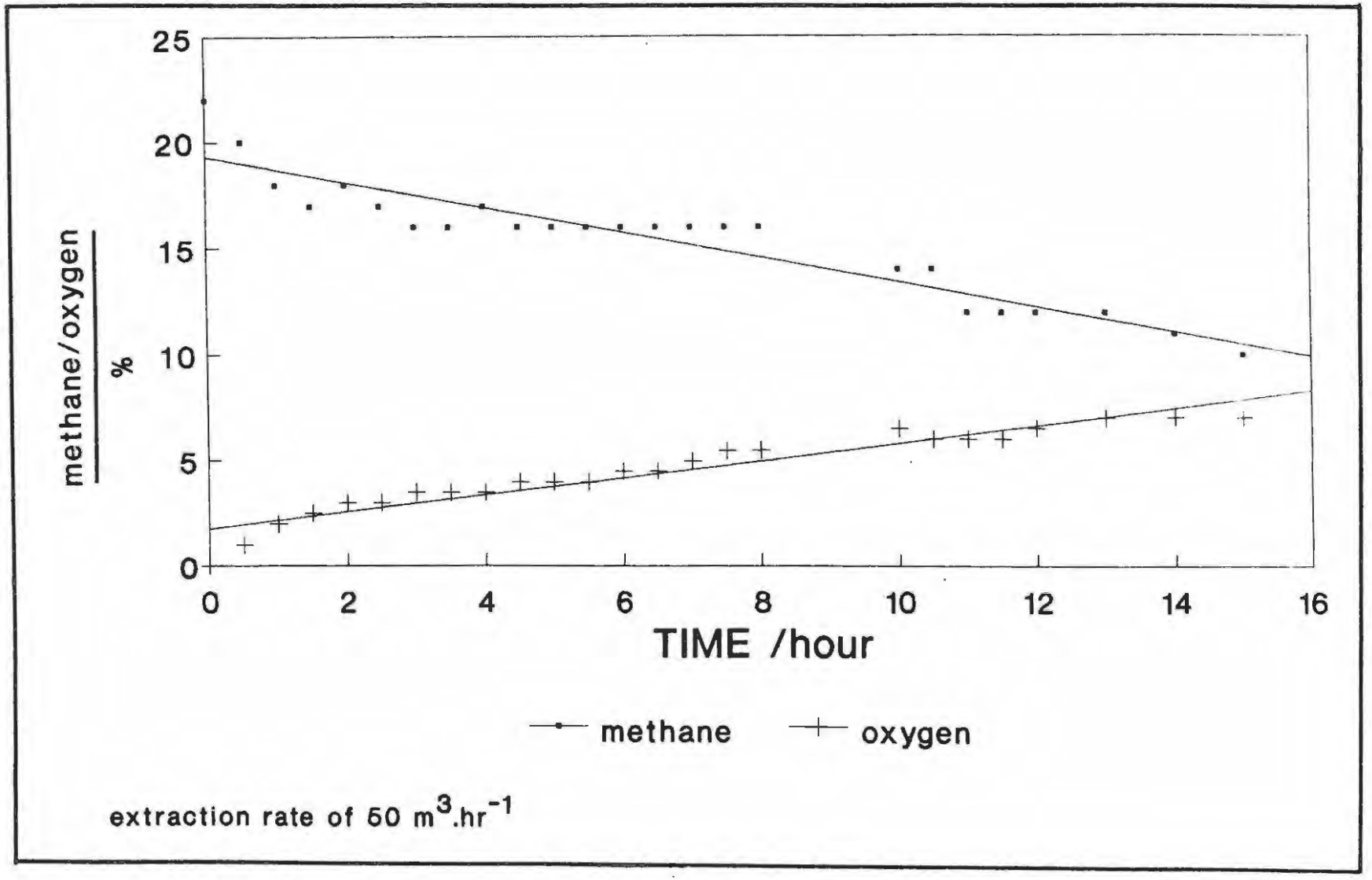

Figure 6. Results of Exhaustive Pumping Experiment carried out on Well A1.

extraction far exceeded the gas generation rate as in the first hour, methane was extracted at 10 $m^3.hr^{-1}$ (50 $m^3.hr^{-1}$ x 20%) and in the 15th hour at only 5 $m^3.hr^{-1}$ (50 $m^3.hr^{-1}$ x 10%). Once pumping was terminated, it took 72 hours for the original gas composition to be regained.

It can be concluded that the methane generation rate is lower than 5 $m^3.hr^{-1}$ and that continuous pumping of LFG at 50 $m^3.hr^{-1}$ is far from desirable if the rehabilitated site is to retain its anaerobicity.

3.2.2 Determination of "Cylinder of Influence":

EPA's Method 2E[1]

This is a somewhat more sophisticated and thorough means of determining the present methane resource than the method discussed in section 3.2.1.

The method is based on the concept of the "cylinder of influence" of a gas extraction well****. The approach is to withdraw gas from a well in the landfill and attempt to identify the volume of refuse from which the gas is being extracted. The application of a vacuum on the extraction well results in a decreased pressure at points in the landfill close to the well. The greater the vacuum, the greater will be the pressure drop. The further the distance from the well, the less the pressure change, until

---

****The term "cylinder of influence" refers to a volume in the landfill site, whereas the more commonly used term "radius of influence" refers only to the radius of the cylinder.

at some distance no change is detected. The point at which this occurs defines the radius of influence for the well in question and the vacuum applied. Definition of this parameter enables both the gas extraction rate and the well spacing of an envisaged extraction system to be determined.

The Environmental Protection Agency (EPA[1]) of the USA has developed a detailed method by which the radius of influence (and hence cylinder of influence) can be ascertained (Method 2E). They suggest that three extraction wells be installed in a cluster of three in a triangular arrangement, as detailed in Figure 7. Sinking three wells, rather than only one, is an attempt to overcome the effects of the heterogeneity of the refuse material. Two types of pressure monitoring probes are then installed. The first type of probe should be located close to each individual well ($\approx$ 3 m) and should penetrate the cover material, but not the refuse itself. These are referred to as shallow probes. The second type of probe (called deep probes) should be installed in the refuse at the level of the perforations of the vertical pipes. Testing is carried out as follows:

(1) The natural flow rate, Q_n, of the gas leaving each well (no pumping) and its methane content is determined. The static pressures (P_i) in the shallow and deep pressure monitoring probes are measured.
(2) The wells are pumped from collectively at twice the natural flow rate ($2Q_n$) for 24 hours.

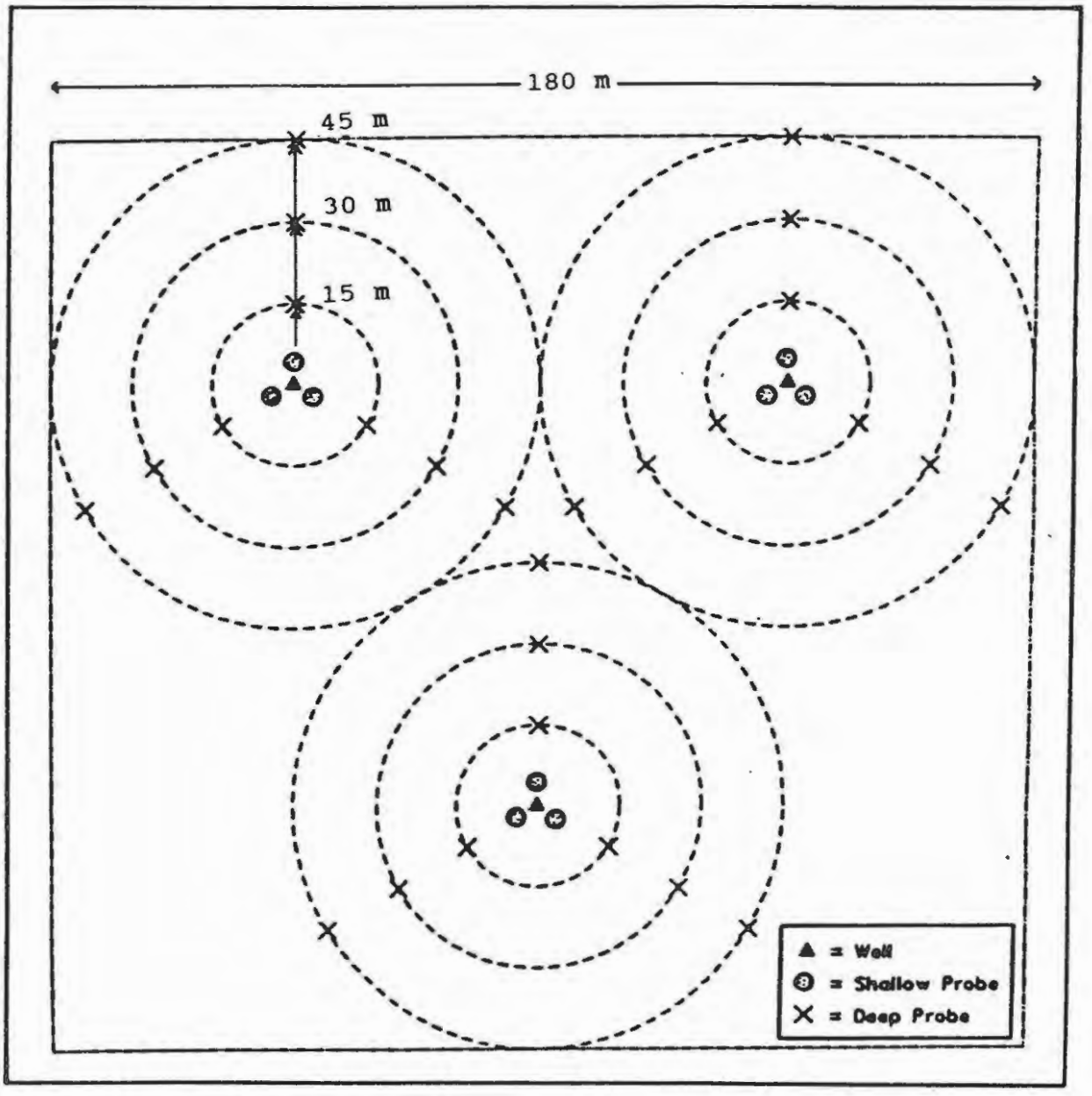

Figure 7. Layout of Extraction Wells and Pressure Probes as Suggested by EPA[1].

(3) To check for air ingress, nitrogen concentrations in the gas are measured and the pressure (P_g) in the shallow monitoring probes is recorded.

(4) If nitrogen concentrations are less than 1% by volume and no negative pressures, (P_i - P_g) < 0.5 cm H_2O or (P_i - Pg) < 50 Pa*****, are detected in the shallow monitoring probes, the pump flow rate is increased to $4Q_n$ and the infiltration tests repeated. The "cut-off" pressure difference of 50 Pa is a reasonable value as the pumping vacuum is usually greater than 500 Pa.

(5) The pumping procedure is repeated at $6Q_n$ etc. until nitrogen concentrations exceed 1% and/or a negative pressure, i.e (P_i - P_g) > 50 Pa, is detected in any one of the shallow monitoring probes.

(6) The pump vacuum is then slowly decreased until nitrogen concentrations are less than 1% and no negative pressures, (P_i - P_g) < 50 Pa, are detected in the shallow probes.

(7) At this flow rate (Q_e), the deep pressure probes are monitored (P_f). The distance from the well at which no pressure difference between the initial static pressure (P_i) and the probe pressure (P_f) exists, i.e. P_i - P_f < 50 Pa, is then the radius of influence (R) for the particular flow rate.

Once the radius of influence is determined, it can be projected onto the whole site and the total number of wells (q) required to extract all the methane from the site can be calculated. The

---

*****1 mm H_2O pressure ≈ 10 Pa

total flow of LFG from the site at time t is then given by qQ_e. The methane flow is mqQe, where m is the percentage of methane in the LFG.

This method does have some deficiencies, primarily related to the concept of a "radius of influence"[2]. Firstly, it is assumed that the gas flow distribution around the extraction well is uniform. This is very unlikely given the heterogeneous nature of refuse material. Secondly, the pressure differences (P_i - P_r) are small and difficult to measure, and the radius may consequently be over or underestimated[2]. Another criticism of the method is that the ingress of air is monitored by a very expensive and time consuming means, namely with nitrogen detecting equipment. It has been suggested that simple and relatively inexpensive oxygen field analysers may serve the same purpose[2]. This does of course assume that oxygen will be drawn through the system before it has time to react with the bacteria. Finally, the method is a fairly complex and very costly one, requiring expensive pressure probes.

3.2.3 A Novel Improvement on EPA's Method 2E[1] Developed at the Grahamstown Landfill Site: Olfaction Method

The EPA[1] method has not been carried out at the Grahamstown Site because of the difficulties of accurately measuring pressure differences ((P_i - P_g) and also (P_i - P_r)) on site. Instead, an inexpensive method based on odour determination has been employed. This method could well be used as a replacement for the pressure measurement technique. Small quantities of a highly

odorous compound replace the pressure probes used in Method 2-E. If the odour of this compound is detected in the gas being pumped out of the well in question within a minute of pumping, it can be assumed that the pump has drawn the odorous gas through the refuse.

Odour experiments were carried out on the rehabilitated section of the Grahamstown Landfill. Thiophene (≈ 1 g) was put down one of the existing extraction wells, well A1 (see Figure 4, Chapter 5) in the rehabilitated site, the well-head sealed, and gas pumped from a different well, well A4 (a distance of 35 m away) at a negative pressure of 2 cm of water. The characteristic smell was detected in well A4 within 60 seconds. This indicates that the extraction wells have a sphere of influence greater than 35 m at the low pumping rate ($12m^3.hr^{-1}$) employed. At higher pumping rates, the sphere of influence would obviously be greater. It is understandable that air ingress was observed during the continuous pumping trials (section 3.2.1) in which gas was extracted at 50 $m^3.hr^{-1}$, as the site boundary is less than 20 m from well A1, so that the sphere of influence extends far beyond the landfilled refuse.

It can be concluded that pumping rates of significantly less than $12m^3.hr^{-1}$ must be used in order to extract gas from the wells in the rehabilitated section of the site if air is not to be pumped into the refuse.

4. COMBINED ASSESSMENT

It has already been mentioned that present generation rates predicted from kinetic modelling should ideally match those determined experimentally. If not, the model can be calibrated/adjusted to the experimental data by equating the rate equation prediction (equation (1)) for present flow rates to those determined experimentally, and applying a correction factor (V) to the parameter $P_{o(j)}$:

$$Q_t = \sum_{j=1}^{j=3} \sum_{t=1}^{t=t} \left(\frac{416 x 0.693 x V x P_{0(j)}}{t_{1/2(j)}} \right) \exp\left(\frac{-0.693 x t}{t_{1/2(j)}} \right) \ldots\ldots (3)$$

This effectively reduces α (which is 416 m^3 methane per dry biodegradable tonne) to a more realistic value. Equation (3) is a one parameter equation and can provide a simple method for predicting the future methane potential of a landfill if the half-life values used by Hoeks are assumed to be correct. Yet another approach to improving the modelling process is to simplify equation (3) by using a composite $t_{1/2}$ value for the three types of refuse material. Equation (3) then becomes:

$$Q_t = \sum_{t=1}^{t=t} \left(\frac{416 x 0.693 x V x P_0}{t_{1/2}} \right) \exp\left(\frac{-0.693 x t}{t_{1/2}} \right) \ldots\ldots (4)$$

It is possible to treat equation (4) as a two parameter (V and $t_{1/2}$) equation which can be solved using at least two values of Q_t. Once these variables are determined, the future methane potential for the site can be accurately predicted. It is

suggested that the flow rate determinations be carried out a year apart, assuming that pumping between the times that the data is taken is never in excess of the natural methane production rate.

This above procedure has not been followed for the Grahamstown Landfill because the site is to small to determine the "cylinder of influence".

5. CONCLUSIONS

The results of flow rate experiments, pumping tests and kinetic modelling for the rehabilitated site are given in Table 4. In conclusion, it must be assumed that the methane generation rate in the rehabilitated section, Q_t, is greater than 0.7 $m^3.hr^{-1}$ (flow rate measurement) and less than 2.9 $m^3.hr^{-1}$ (upper estimate from kinetic modelling).

TABLE 4
Determination of Methane Generation Rates for the Rehabilitated Site

SECTION	METHOD	METHANE FLOW - JULY 1991 $m^3.hr^{-1}$
2.2	kinetic modelling (ideal estimate)	2.5
2.2	kinetic modelling (high estimate)	2.9
2.2	kinetic modelling (low estimate)	0.5
3.1	flow rate measurement, Q_n	0.7
3.2.1	continuous pumping	<5
3.2.3	odour tests	<4

In the operational site, it is estimated that the methane generation rate is greater than 44 $m^3.hr^{-1}$ (flow rate measurements) and less than 150 $m^3.hr^{-1}$ (upper estimate from kinetic modelling). If LFG is to be extracted from the entire site, it is suggested that extraction rates be of the order of 100 $m^3.hr^{-1}$ and that they not exceed 200 $m^3.hr^{-1}$.

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CHAPTER 11

LANDFILL GAS UTILISATION

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1. INTRODUCTION

The combustion of methane in air (equation (1)) is exothermic, the reaction producing 890 kJ of energy for every mole of methane that is burnt.

$$CH_4 + 2O_2 \rightarrow CO_2 + 2H_2O \qquad \Delta H^\circ = -890\ kJ.mol^{-1} \qquad (1)$$

It is easily calculated that methane has an energy value of 36 MJ per m^3 at STP (see Chapter 5, section 2). This is approximately the same amount of energy that is available in one litre of petrol. The energy value of LFG is of course lower than this, the amount depending on gas composition.

The world's first commercial LFG extraction facility was developed in 1975 in the USA[20]. During 1990, 5 000 million m^3 of LFG (containing 57% methane on average) was exploited from 453 plants throughout the world[10]. This gas had a total energy content of 102 000 TJ[10], which is equivalent to 28 000 kWh (assuming 100% conversion to electricity), or 3 mtce or 2 mtoe* (see Table 1).

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*mtce = million tonnes coal equivalent
mtoe = million tonnes oil equivalent

TABLE 1

LFG Extraction Plants Throughout the World
(Adapted from Gendebien *et al*[9])

COUNTRY	NUMBER OF LFG PLANTS	LFG EXPLOITED 10^6 $m^3.yr^{-1}$	ENERGY 10^3 $TJ.yr^{-1}$
European Economic Community			
Belgium	1	0.127	0.0025
Denmark	6	8.5	0.167
Fed. Rep. of Germany	98	400	7.9
France	7	50	1.0
Greece	0	-	-
Ireland	0	-	-
Italy	13	38	0.75
Netherlands	10	80	1.6
Portugal	0	-	-
Spain	2	?	?
United Kingdom	26	178	3.5
TOTAL EEC	163	755	14.9
Other European Countries			
Austria	1	1.5	0.03
Sweden	12	60	1.2
Switzerland	7	9	0.178
Turkey	1	?	?
TOTAL	21		
America			
Brazil	6	?	?
Canada	9	26	0.5
Chili	1	?	?
United States	243	4 300	85
TOTAL	259		
Africa			
South Africa	2	?	?
Australia	4	?	?
Asia			
Hong Kong	1	?	?
India	3	?	?
GENERAL TOTAL	**453**	**5152**	**102**

Utilisation options for the methane component of LFG can be categorised as follows[2,31]:

- DIRECT USE of LFG without extensive pretreatment, primarily for heating purposes.
- POWER GENERATION using gas engines or turbines, with or without some pretreatment of the LFG.
- EXTENSIVE GAS CLEANUP to produce a higher quality fuel or chemical feedstock.

In addition to methane recovery, the recovery of CO_2 from LFG is possible, although the author is not aware of any such documented cases. CO_2 is used in carbonation, refrigeration, aerosol packaging and as a neutralising agent for pH control[36]. If LFG is purified by the removal of CO_2, the absorbed/adsorbed CO_2 might prove worthwhile extracting.

2. LFG PRETREATMENT

The extent to which LFG is pretreated before utilisation is determined by its end use. Pretreatment steps include removal of condensate/dewatering, cooling, particulate removal, compression, removal of CO_2, removal of trace organics and removal of H_2S[36]. These purification measures are largely (except possibly for CO_2 removal) related to the potentially corrosive properties of LFG.

2.1 Corrosion Aspects

In addition to being saturated with water, LFG contains a multitude of trace organic components (see Appendix 1), including mercaptans and halocarbons and possibly significant quantities of hydrogen sulphide (if gypsum or sewage sludge have been deposited)[6]. The generation of acids from these compounds (eg. hydrochloric acid from chlorocarbons and sulphuric acid from mercaptans and hydrogen sulphide) is the cause of corrosion. There are many documented cases of enhanced chemical corrosion in engines due to the use of raw LFG as a fuel[31].

At the Braunschweig LFG plant in Germany, an internal combustion engine is documented to have broken down completely due to corrosion[6,37]. This was ascertained to have resulted from the large quantities of aliphatic chlorocarbons in the LFG from the landfill (which co-disposes municipal and industrial waste)[6].

Corrosion of boilers which use LFG directly, without any pretreatment, has been known to occur[6].

2.2 Pretreatment Processes

A number of processes have been developed for the pretreatment of LFG prior to utilisation, in order to reduce corrosion damage. It is in any event advisable to remove these pollutants before utilisation in order to reduce potential emissions[6].

2.2.1 Condensate Removal/Dewatering and Particulate Removal

Regardless of the energy application, condensate and, if necessary, particulate removal are considered to be essential pretreatment steps[36]. Much of the condensate (see Chapter 5) can easily be removed in a simple knock-out pot comprised of baffle plates, as is done at the Grahamstown Landfill LFG Plant. A higher degree of dewatering can be achieved with the use of chiller/refrigeration (0-5 °C) units. This is practised at the Stewartby Landfill, UK[14] and at the Robinson Deep Landfill in Johannesburg, South Africa. Absorption methods such as glycol dehydration (used at Tilburg, Netherlands[35]) and adsorption systems, which commonly employ silica gels, are also used for extensive dewatering[36].

LFG is usually free of particulates, but if not, these particulates must be removed prior to entering the extraction pump/blower/compressor[6].

2.2.2 Removal of CO_2

If it is required that the heating value and/or purity of LFG be upgraded, it is necessary to remove carbon dioxide. CO_2 can be removed by absorption into a liquid, adsorption on a solid, membrane separation, cryogenic separation or chemical conversion to another compound[36].

Liquid absorption is by far the simplest method, especially if

water is used as the solvent (equations (2) and (3)). The process, known as water scrubbing, involves CO_2 absorption at pressures of approximately 10 atmospheres and is the most attractive process technically and financially[35].

$$CO_2 \xrightarrow{H_2O} CO_2(aq) \qquad (2)$$

$$CO_2(aq) \rightleftharpoons HCO_3^- + H^+ \qquad (3)$$

Another process, the Krysol Process, which uses methanol as the physical solvent, incorporates the subsequent desorption of CO_2 for the production of food grade liquid CO_2[21]. Methanol is, however, not as efficient a solvent for CO_2 as is water; the solubility of CO_2 is 171 g in 100 ml water and only 31 g in 100 ml alcohol at STP[39].

Adsorption processes usually entail the simultaneous removal of H_2S, moisture and other impurities depending on the adsorbent. Adsorbents include activated carbon, alumina, silica and silicates, while molecular sieve processes for the removal of CO_2 have also been developed[36].

Membrane separation, based on the relative permeability of gases, (extensively reviewed by Rautenbach and Ehresmann, 1990[29] and van Bladeren, 1989[38]) has been commercially developed by the Monsanto Company[23].

2.2.3 Removal of Hydrogen Sulphide and Hydrocarbons

Activated carbon is extensively used in the purification of LFG for the removal of hydrogen sulphide and/or trace organic components[6,28]. The use of activated carbon for this purpose is related to its high surface area and affinity for certain, mainly organic, compounds. The design of such a system must make allowance for the fact that different hydrocarbon compounds exhibit different adsorption behaviours, and that as a result of competitive adsorption, some halogenated hydrocarbons might be displaced by alkanes and aromatics, which are preferentially adsorbed[28]. Once the adsorption columns are fully loaded, they can be regenerated by desorption.

Hydrogen sulphide and mercaptans can also be removed by adsorption on iron oxide[35].

A catalytic process for the removal of halocarbons from LFG has been investigated. The process involves the transformation of halocarbons by catalytic dehydrohalogenation into hydrocarbons at 300-600 °C[6].

3. UTILISATION OPTIONS

The decision as to what the LFG will be used for is ultimately dictated by the proximity of potential consumers. The low energy density of LFG makes transport of the gas expensive, even in its purified state. It is also preferable that any industries that

are supplied have a sustained energy demand.

3.1 Direct Use

In the results of a survey of 242 LFG plants worldwide, 36% of the plants are positively identified as using LFG directly, without any purification other than dewatering, for the production of heat[30].

Direct use of the gas necessitates the close proximity (<2 km)of the consumer to the landfill site in order to be economically viable. If there are no consumers in close proximity, it may be better to use the gas for power generation[31].

Direct applications include the following:

- The heating of WATER BOILERS for the production of hot water and/or steam. The steam may be fed to a gas turbine for electricity generation. At Viborg Kommune, Denmark, LFG is transported a distance of 2.5 km and burned in a 2.6 $GJ.hr^{-1}$ boiler, producing hot water for the local district heating grid[9].
- Use as a boiler fuel for the heating of GLASSHOUSES. The Bilham Quarry Landfill in Yorkshire, UK, which is a small site containing some 100 000 tonnes of refuse, runs an extraction plant which pipes gas a distance of 250 m to a boiler which heats a glasshouse (21-23°C) used for growing geraniums, cyclamens and peppers. The consumption of LFG is

approximately 175 000 m^3 per annum. Moisture is removed from the pipeline by a simple knock-out pot[4].

- SPACE HEATING. Examples include district heating in Austria and Sweden[37].
- FIRING OF KILNS/FURNACES/INCINERATORS. In the UK, there are 10 plants which use LFG for the firing of brick kilns[7]. At the Stone Pit site in Kent, the kiln used for cement production runs on 50% LFG. The manufacturers prefer LFG because it leaves no ash when burnt[33]. The Communa di Modena landfill in Italy supplies a toxic waste incinerator at the site with 9 000 m^3 of LFG per day[9].
- The DRYING of bitumen in a Tarmac Coating Plant is fuelled by LFG from the Allsopps Hill landfill, a distance of 750 m away[11].
- Use as a COOKING FUEL. A community kitchen in Brazil utilises LFG from a landfill which receives 500 tonnes of waste per day[24].
- LIGHTING. A site in Delhi, India, reportedly pipes LFG a distance of 1 km for use as a fuel for domestic lighting, using conventional thorium heating mantles, cooking and heating (pers. comm. T.M. Letcher).

It should be mentioned that biogas from the anaerobic treatment of sewage can be (and is) used for many of these applications, as discussed in Appendix 4.

3.2 Power Generation

The generation of electric power is the most widely practised utilisation option on a world-wide scale. In the study mentioned previously, 55% of 242 world-wide LFG utilisation projects were classified as producing electric power[30]. There are 18 such projects in the UK[7] and 55 in the USA[10]. In the USA this is generally the preferred utilisation option (70%) because of the absence of potential users in close proximity to the sites[40]. Electricity generation generally proves to be a viable proposition if there is no demand for gas close to the site.

Power may be generated with the use of spark-ignition engines, gas turbines and dual fuel engines[30]. Some characteristics of these electricity generating machines are given in Table 2. The LFG is generally used directly, without any pretreatment other than liquid removal and compression, extensive pretreatment not being required[14].

TABLE 2
Characteristics of Electricity Generating Machines
(Adapted from Moss, 1991[24])

MACHINE TYPE	TYPICAL EFFICIENCY %	TYPICAL LIFE hours	TYPICAL SIZE MW
spark-ignition	28-30	40 000	< 0.5
dual fuel	35-40	150 000	1 - 10
gas turbine	26-27	60 000	> 2
steam turbine	15-20	150 000	> 2

3.2.1 Spark-Ignition-Engines

The Stewartby Landfill in Bedfordshire was the first site to produce electrical power from LFG in the UK[25]. Three 275 kW Dorman spark-ignition reciprocating engines (which start directly on LFG) generate electricity for export to the national grid[14]. Apart from condensate removal and compression, the gas is not pretreated. The lubricating oil in the engines has to be changed regularly as it has been found that it acidifies readily. A positive correlation between halogenated hydrocarbon concentration and oil acidity has been established at the site. No major corrosion problems have however been detected. Minor problems include the wearing of valves, lacquer deposits on pistons and silica contamination of the lubricating oil[14].

Spark-ignition engines have capacities ranging from 3 to 1 000 kW[17] and are run off LFG with a methane content as low as 30%[33] and even 25%[22]. Typical engine efficiencies range between 28 and 30%[25]. Efficiency increases to 80-85% if the waste heat is recovered[37]. In Denmark, a 6 kW combined heat and power (CHP) unit runs off LFG with a methane content of 42%[9].

The most significant environmental impact of power generation is the emission of pollutants to the atmosphere[14]. While trace components are significantly removed as a result of the dewatering and compression of gas prior to power generation, if carburation is not carefully set and adjusted, elevated levels of carbon monoxide, unburned hydrocarbons, NO_x and particulates

may be emitted in the exhaust gases[41]. At the Stewartby site, levels of particulates, carbon monoxide and NO_x have at times been higher than EC (European Community) limits. Dioxin emissions are only just above levels of detection[14].

3.2.2 Gas Turbines

Gas turbines operate on a far larger scale than do gas engines, and sizes are typically above 2 MW[25]. The turbine at Packington, UK, is rated at 3.7 MW and has effectively transformed the landfill into a power station generating enough electricity to supply the energy requirements of approximately 5 000 houses[2].

Advantages of gas turbines over the more commonly used reciprocating engines include the following[2]:

- CO_2, which is normally a waste product of energy conversion, is used by the turbine to produce electricity, as turbine operation entails the utilisation of the expansive properties of all the gases in a gas mixture.
- The high combustion temperatures used significantly reduces the potential emission of pollutants. The gas turbine at Packington, UK, emits pollutants (CO, SO_2, HCL, NO_x, particulates, hydrocarbons, dioxins, furans) at levels within EC limits.

The relative disadvantages of gas engines include:

- high capital cost[14].

- the need for high compression, as high fuel injection pressures are required for turbines[14].
- a slightly lower efficiency, 26-27%[25].

The running of the gas turbine at Packington, which can be directly started on LFG, has not been without minor problems, but none of these problems appear to relate to the use of LFG as a fuel[14].

3.2.3 Dual Fuel Engines

Dual fuel engines are converted diesel engines which use 10-20% diesel to produce ignition of the diesel-gas mixture[17]. They operate at higher efficiencies than either gas turbines or spark-ignition engines, 35-40%[25]. Typical sizes range between 1 and 10 MW[25]. Two dual fuel engines, capable of 2 MW each, run off LFG from the Brogborough Landfill, UK[25]. This type of engine is not as economical to run as the newer spark-ignition engine and is therefore not used as extensively[25].

3.3 Extensive Gas Purification

Most LFG applications are related to heat and power generation, which do not require extensive gas pretreatment. Injection of LFG into utility pipelines, use as a vehicle fuel and use as a chemical feedstock require that the gas be extensively pretreated. Only 9% of 242 sites are reported to upgrade LFG to a higher quality fuel[30], gas cleanup to produce methane gas of

natural gas quality (>90% CH_4) being both complex and expensive[31].

It has been determined that while electricity generation can be financially viable at all capacities, extensive upgrading of LFG below a gas production of 250 $m^3.hr^{-1}$ is not viable[35]. At LFG extraction rates of 750 $m^3.hr^{-1}$, electricity generation and upgrading break even financially[35].

3.3.1 Pipeline Gas

At a number of large landfills in the US, LFG is upgraded to natural gas quality (>90% CH_4) and injected into utility pipelines[37]. An example of such a scheme is the one at Palos Verdes Landfill, California. The process entails the cooling of the gas, removal of moisture, hydrocarbons, other contaminants and CO_2 by silica gel, activated carbon and molecular sieves and compression to pipeline requirements[36].

A common process used for the upgrading of LFG in the US is the Selexol Process. A solvent, dimethyl ether of polyethylene glycol, is used to physically absorb bulk quantities of CO_2 and hydrocarbons, particularly sulphur-based compounds, while simultaneously dehydrating the gas[36].

In contrast to the complex and very expensive process upgrading projects in the US, a scheme in Santiago, Chile, where LFG is used to <u>supplement</u> a town's gas distribution facility, runs on

raw, unscrubbed LFG. No major problems have as yet been documented[31].

3.3.2 Vehicle Fuel

Methane is used as a fuel in over 700 000 vehicles world-wide, in 38 countries[34]. LFG once scrubbed of CO_2 and other unwanted trace gases, and compressed into cylinders, can therefore replace petrol. Unfortunately, extremely high pressures are required to liquefy methane and as a result methane can only be transported as a gas in heavy gas cylinders, resulting in a limited range for vehicles of about 100 km. This makes the purification of LFG for use as a vehicle fuel economically marginal[31]. Brazil and Italy are the only countries documented as using LFG in this manner[31]. The gas is compressed to approximately 200 atmospheres after CO_2 and moisture removal[24].

The advantages of using methane as opposed to conventional petrol include[3]:

- SAFETY. Methane (molar mass of 16 $g.mol^{-1}$) has a density of 16/25 or 0.64 $kg.m^{-3}$ at STP and is lighter than air which has a density of 1.2 $kg.m^{-3}$ at STP. In the case of gas leakage, methane will therefore dissipate readily into the atmosphere. (According to Grahams' Law of Diffusion, methane diffuses $\sqrt{30/16}$ or 1.4 times faster than air). The ignition temperature of methane (octane rating 130) 700°C, is considerably higher than that of petrol, 420°C.
- POLLUTION. Exhaust emissions of CO, CO_2 and unburned

hydrocarbons are considerably lower than for petrol.

- PERFORMANCE. The higher octane rating of 130 allows for more efficient operation. The cleaner burning fuel doubles engine life.

Disadvantages relate to the space required for cylinder storage, the dangers inherent in transporting heavy gas cylinders which are under high pressure (200 atmospheres) and the limited distances that can be covered.

3.3.3 Methane Conversion

Methane is thermodynamically stable with respect to its elements so that pyrolytic reactions to make other hydrocarbons have unfavourable energies of reaction. The energy of the C-H bond, 440 $kJ.mol^{-1}$, is far higher than most other bonds and methane is hence very stable and unreactive[26].

A mere 7% of natural gas (>90% CH_4) in the UK is used to make synthesis gas ($CO + H_2$) for the production of ammonia or methanol[26]. The process is very expensive, and hence the rather small scale of the operation. The use of LFG for this same purpose is unlikely to be economically feasible due to the extensive gas cleanup that will be required.

It therefore comes as no great surprise that, apart from one case in South Africa, LFG is not used as a chemical feedstock. At the Robinson Deep Landfill Site in Johannesburg, South Africa, LFG

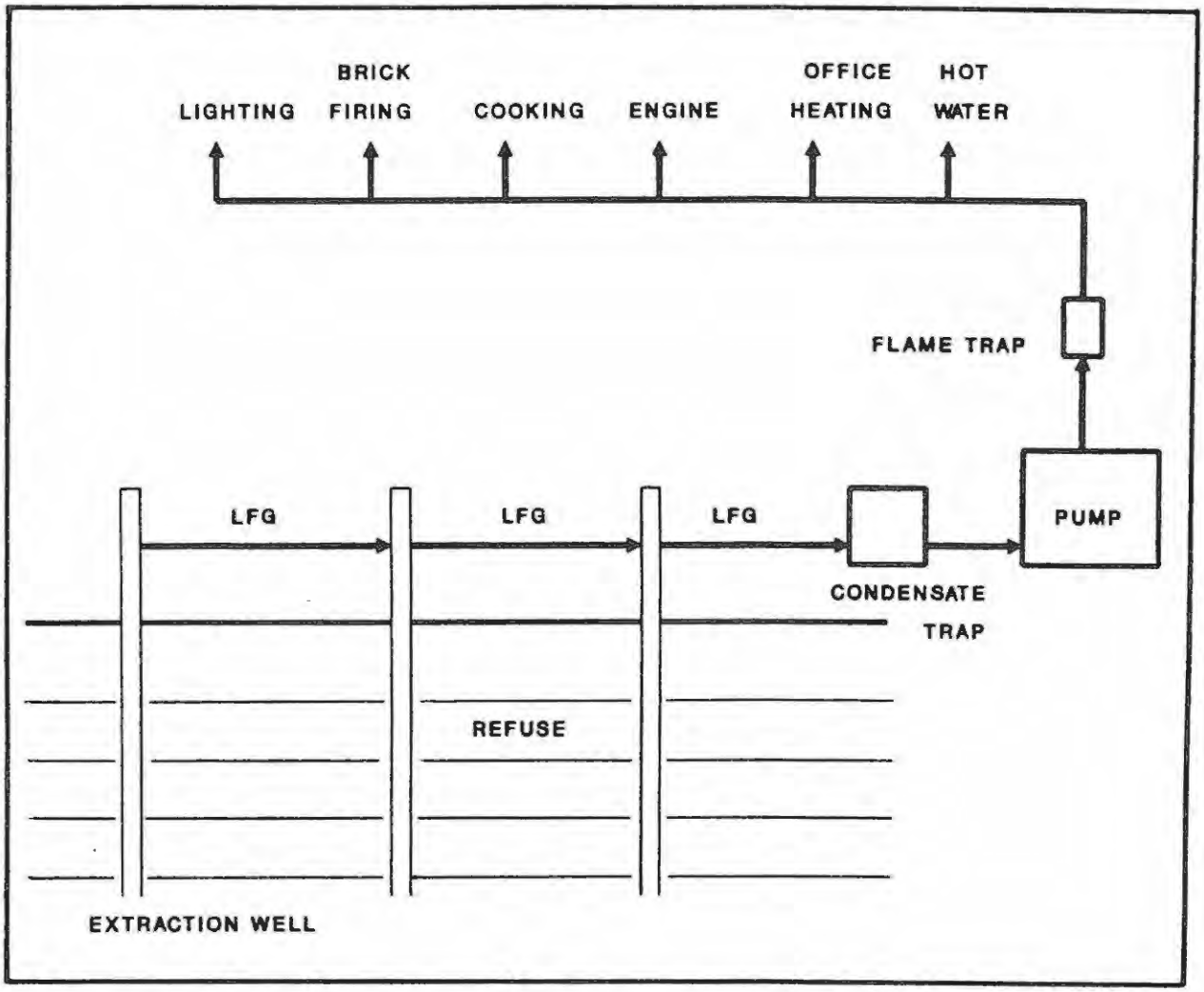
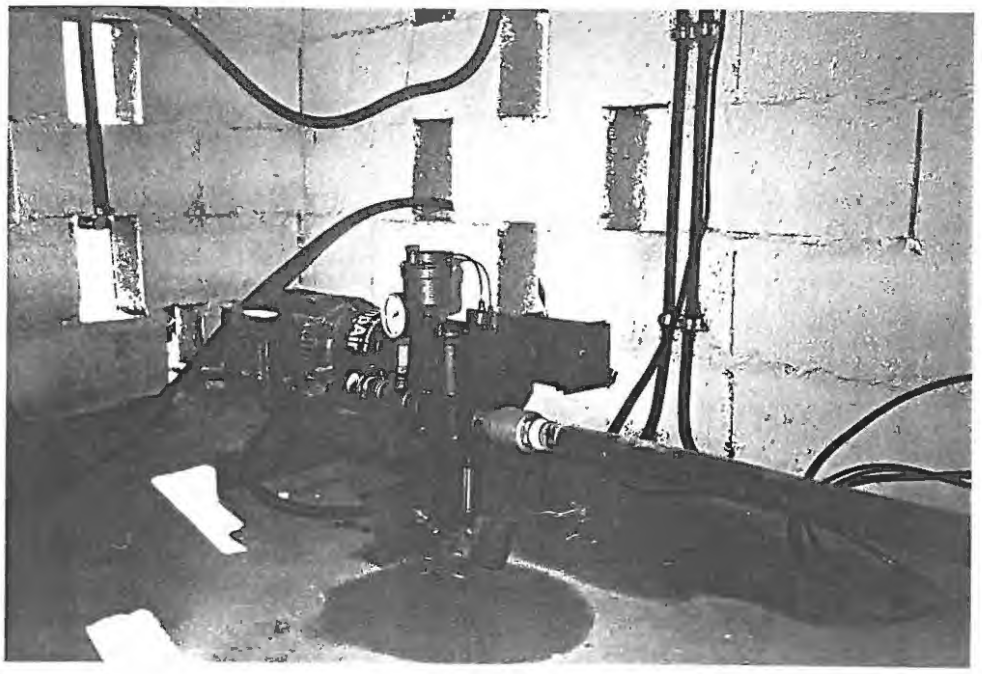


Figure 2. The Grahamstown LFG Plant

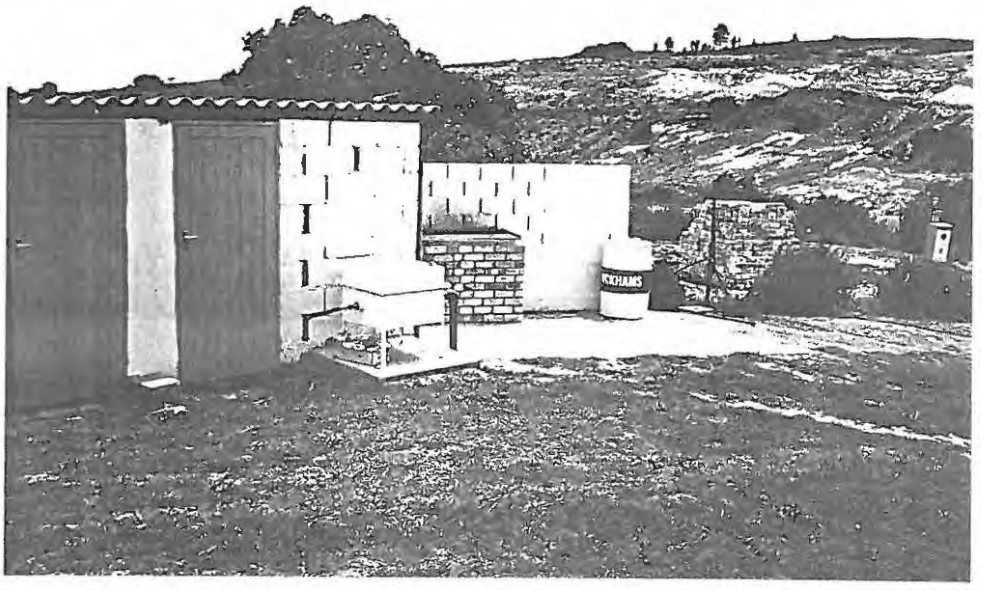
extracted with a REAVELL rotary vacuum pump/blower which has a capacity of 60 $m^3.hr^{-1}$ and a vacuum capability of 500 mm gauge (Photograph 1). Before entering the pump, the gas first passes through a condensate trap, which consists of a series of baffle plates, in order to cool and dewater it (Photograph 2). A flame trap for the prevention of flashbacks is installed in the gas line between the pump and the manifold (Photograph 3), from where the gas is directed to the various utilisation options.

4.1 Hot Water System

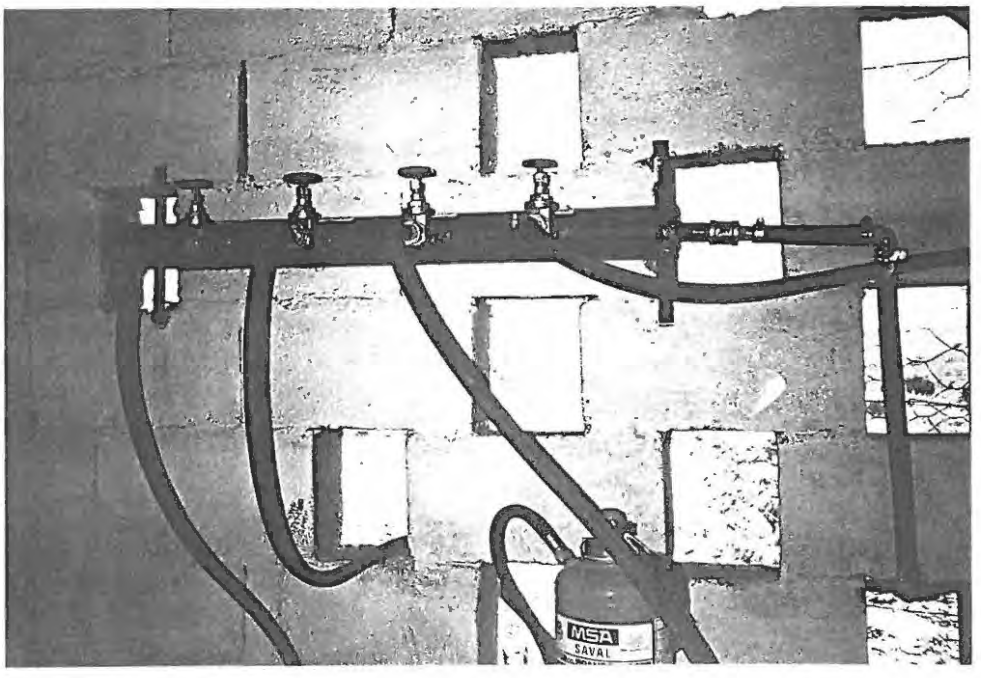
The gas is used to provide hot water at two locations: (i) at the offices on the site, a distance of 10 m from the pump house and (ii) at the landfill caretaker's cottage, a distance of 300 m from the pump house (see Figure 4, Chapter 5). In both instances, gas is extracted at the rate of 50 $m^3.hr^{-1}$ (the suction on the pump cannot be varied) and approximately a quarter of the gas, 12 $m^3.hr^{-1}$, (which contains approximately 35% CH_4, 25% CO_2 and 2% O_2, by volume) is piped to each gas "geyser". The burners consist of a tin can filled with stones in the first case, and an adapted burner from a conventional gas geyser in the second. This burner heats up a 10 litre steel container which is connected to a 180 litre storage drum 150 cm above it. Cold water from the municipal mains fills the storage tank to a level determined by a ball valve and also fills the smaller container used for heating the water via the copper tubing linking the two vessels. Upon heating, the hot water moves from the small tank to the larger by convection. In this manner, all the water in the large tank



Photograph 1. The REAVELL Vacuum Pump used to Extract LFG at the Grahamstown Landfill Site



Photograph 2. The Pumphouse with the Condensate Trap in the Foreground.



Photograph 3. The Manifold from which LFG is Directed to the Various Utilisation Options.

is eventually heated. Both tanks, and all the piping, are insulated with a glass fibre based material. (See Photograph 4)

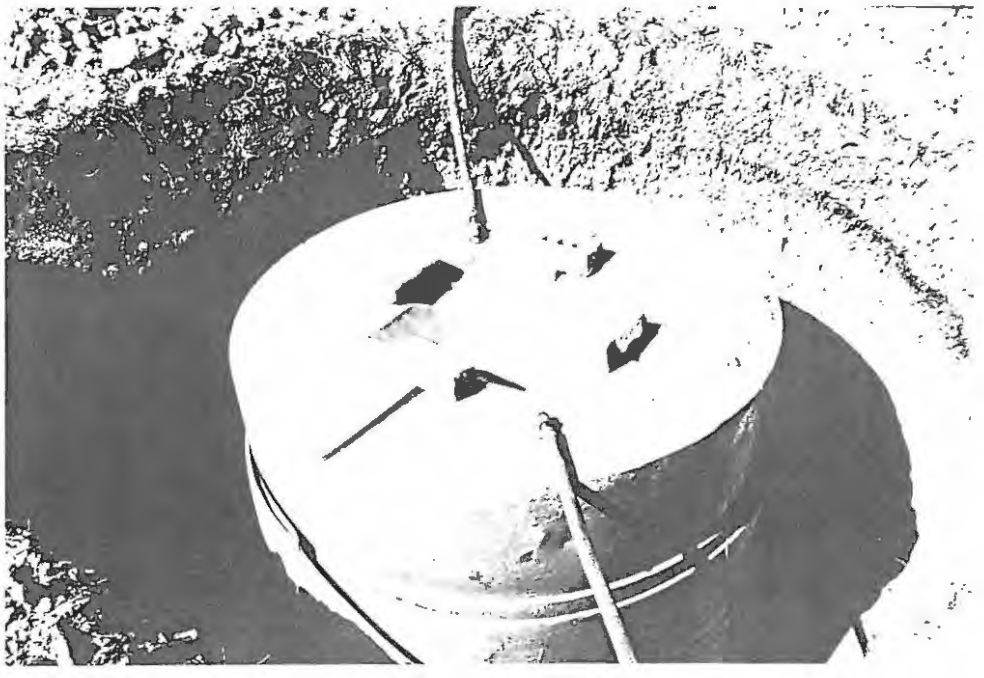
The staff at the landfill can use the hot water for taking showers, within only 60 minutes of lighting the gas supply.

A gasometer (Photograph 5), which is as yet not fully operational, has recently been installed at the site, a distance of 15 m from the pumphouse. Its purpose is the provision of a storage facility for the gas, so that the suction pump does not have to be switched on whenever gas is required. The unit consists of a slightly tapered cylindrical PVC tank (diameter 1.5 m, height 3 m) which has been cut in half and the slightly smaller half placed inside the larger, as shown in Figure 3. Water fills the lower 120 cm of the bottom tank. As gas is pumped into the top half of the tank, it rises, with the water providing a seal. The tank has a gas storage capacity of approximately 1-1.5 m^3. Problems that have been experienced include (i) hindering of the rising of the top half of the tank by the not entirely cylindrical nature of the soft PVC tank and (ii) the development of an inadequate gas pressure in the tank. It is intended to solve the first problem by attaching guides. The second has been rectified by weighting the top of the tank with bricks weighing 150 kg.

4.2 Office Heating

LFG is piped directly, at the rate of 10 $m^3.hr^{-1}$, to a gas burner

Photograph 4. The Hot Water "Geyser".



Photograph 5. Initial Testing of the Gasometer.

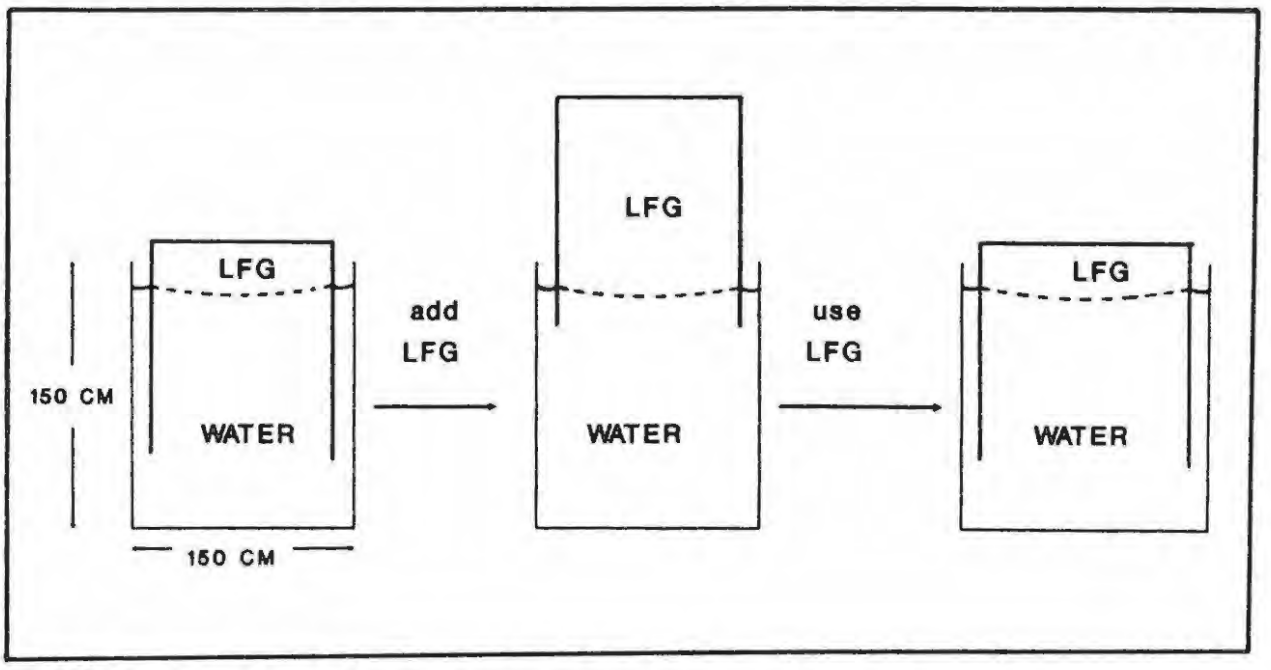


Figure 3. Operation of the Gasometer at the Grahamstown LFG Plant.

in the office at the site. The office can thus be heated during the winter months and water can be boiled for the preparation of hot drinks (Photograph 6).

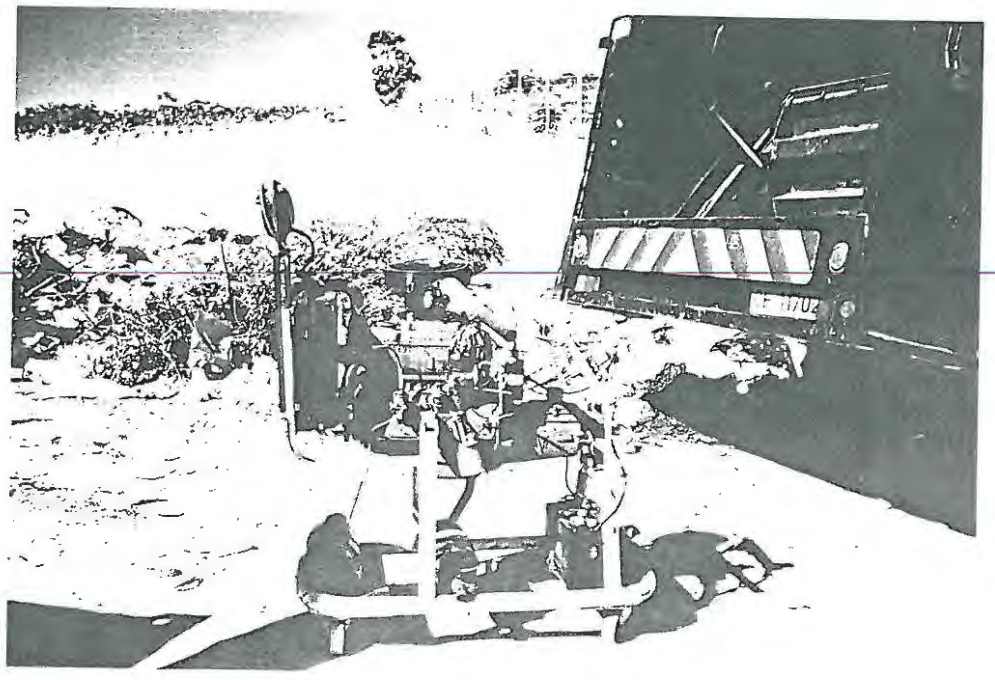
4.3 Gas Engine

A 1 600 Opel motorcar petrol engine runs directly off the LFG from the site (Photograph 7). The engine is started on petrol and gradually switched over to LFG while simultaneously adjusting the air intake. The LFG is fed directly into the carburettor, through the air cleaner. No adjustments have been made to the engine apart from attaching the necessary fittings to the carburettor for the LFG supply. The engine does not as yet serve any purpose, apart from demonstrating that conventional petrol engines can run directly on LFG.

4.4 Clay drying

The feasibility of drying the kaolin mined at a nearby clay mine with LFG has been investigated. In order to completely dry 4 tonnes of clay in one hour, i.e. remove 10% water by mass, approximately $1x10^6$ kJ of energy is required per hour. From various assessments of the quantity of LFG available from the Grahamstown Landfill (Chapter 10), it has been estimated that 100 m^3 of LFG can be extracted from the whole site every hour. This amounts to 40 $m^3.hr^{-1}$ methane (methane content of 40% by volume), or $1.4x10^6$ kJ of energy per hour. The site should therefore be capable of supplying the energy required for the

Photograph 6. The Office Heater.



Photograph 7. The Opel 1600 Engine which Runs on LFG.

drying of the clay, taking a heating efficiency of 70% into account.

4.5 Lighting

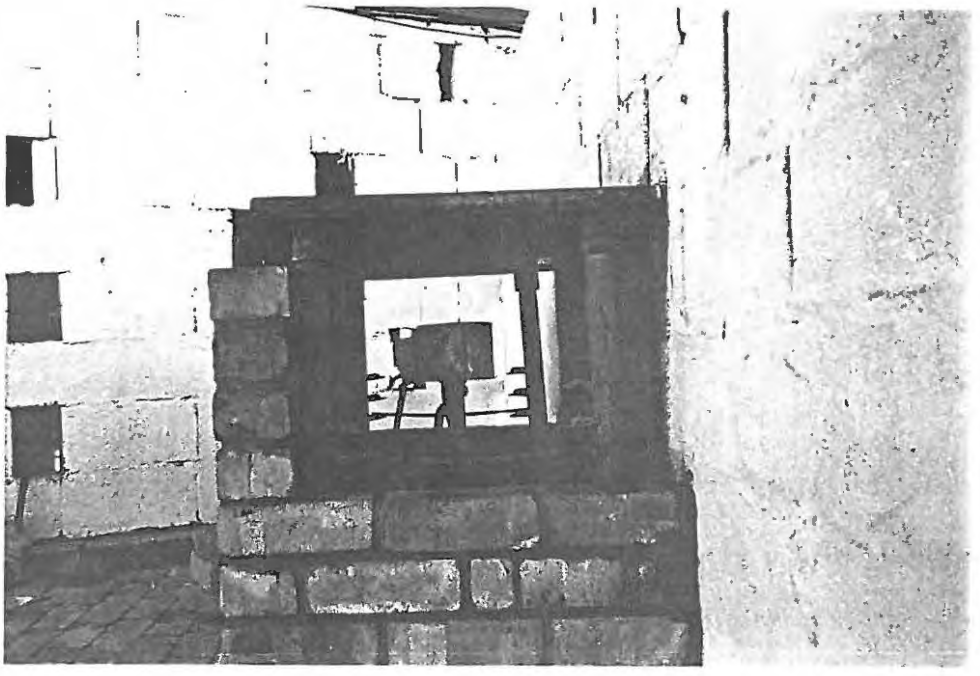
Gas lighting using LFG as the fuel for conventional thorium heating mantles, which normally burn on L.P.G., has been investigated with a measured degree of success. Once the jets controlling gas flow are removed, the mantles do burn, but not as efficiently as on L.P.G.. A gas lamp specially designed for operation on low pressure gas has also been investigated, but with no success.

4.6 Brick Kiln

Initial LFG utilisation experiments at the Grahamstown Landfill involved the construction of a brick kiln. LFG was pumped into the kiln at a rate of 5 $m^3.hr^{-1}$ for 10 days. Of the 5 000 bricks, only 1000 were observed to be well baked[17].

4.7 Cooking

Visitors to the site are regularly treated to meals cooked with LFG. The barbecue facilities consist of a large metal plate heated by tin can burners (Photograph 8).



Photograph 8. The Barbecue Facility.

5. ECONOMICS

5.1 The Energy Available from Landfill

The following analysis of the amount of energy potentially available from landfilled waste is based on the per capita rate of solid waste production. The results of a technical investigation into per capita refuse production are given below[15]:

INCOME STATUS OF COUNTRY	WASTE PRODUCTION kg.cap^{-1}.day^{-1}**
low-income countries	0.4-0.6
middle-income countries	0.5-0.9
industrialised countries	0.7-1.8

A figure of 1 kg.cap^{-1}.day^{-1}[16] is used in the following analysis, in which it is assumed that all of this waste is landfilled. The analysis is as follows:

- The results of the refuse survey conducted at the Grahamstown Landfill (Chapter 8) indicate that the domestic refuse disposed of has an average moisture content of 30% and that only 53% of this dry waste is biodegradable. The amount of dry biodegradable material thus available for methane production is :

**kg.cap^{-1}.day^{-1} refers to the rate of waste production in kilograms per capita per day.

0.365 x 0.7 x 0.53 or 0.135 tonne.cap^{-1}.yr^{-1}.***

- Ideally, each tonne of dry biodegradable waste produces 416 m^3 of methane (Chapter 9). Hence maximum possible methane production is:
  0.365 x 0.7 x 0.53 x 416 or 56 m^3.cap^{-1}.yr^{-1}.
- A cubic metre of methane, at STP, contains 36 MJ of energy (Chapter 3). Energy production is thus:
  0.365 x 0.7 x 0.53 x 416 x 36 or 2 028 MJ.cap^{-1}.yr^{-1},
  which is equivalent to 563 kW hr.cap^{-1}.yr^{-1} or 64 Watt.cap^{-1}.
- Considering that domestic electricity consumption averages 300 Watt.cap^{-1}[16], which is equivalent to an energy consumption of 9 461 MJ.cap^{-1}.yr^{-1}, landfill can <u>potentially</u> supply (2 028/9 461)% or 21% of a city's energy requirements. This is an <u>ideal</u>, and therefore unrealistic, estimate.
- The conditions for methane production in a landfill are never ideal and approximately only 100 m^3 of methane are produced from every dry tonne of biodegradable refuse. (In an assessment of LFG production in the UK, a figure of 111 m^3 CH_4.tonne^{-1} is used[5]). It must also be remembered that the above analysis assumes that all the methane produced is collected. If only 50% of the methane is in fact collected (a reasonable assumption), it is calculated that LFG can only supply 2.5% of a city's domestic energy requirements. This is energy that can be used directly, without conversion to another form, such as electricity.

***tonne.cap^{-1}.yr^{-1} refers to the rate of waste production in tonne per capita per year.

- As regards electricity production from LFG, it must be taken into account that the engines used to produce electricity only run at 30% efficiency. Only 0.8% of a city's domestic electrical energy demand can therefore be supplied by LFG.

5.2 The Cost of Energy from Landfill

Even though landfill can provide only a small amount of the energy required by a city, the feasibility of using energy from landfill can only be assessed once a detailed cost analysis is carried out. This cost is very dependent on the number of people in a community that are served by the landfill concerned.

5.2.1 Cost of the LFG Extraction Plant

The plant considered in this analysis is one in which LFG is extracted at a rate of 50-200 $m^3.hr^{-1}$ from a relatively small site serving 100 000 people. The costing is based on experience at the Grahamstown Landfill Site which serves approximately 80 000 people.

	COST SOUTH AFRICAN RANDS
drilling costs (10 holes)	10 000
piping	12 000
fittings (valves, pipe connections etc.)	5 000
condensate and flame traps, flow meter etc.	5 000
extraction pump/blower (0-250 $m^3.hr^{-1}$)	12 000
standby pump	12 000
labour (installation)	4 000
CAPITAL COST	**60 000**

5.2.2 Cost of LFG Utilisation

(i) Direct Utilisation - Heating

	COST SOUTH AFRICAN RANDS
extraction plant	60 000
hot water system (2x200 l boilers/geysers)	12 000
CAPITAL COST	72 000
annual redemption (over 5 years, 20% pa.)	25 000
operating costs and repairs	15 000
TOTAL ANNUAL COST	**40 000**

Consider that LFG, with a methane content of 40%, is extracted at a rate of 200 $m^3.hr^{-1}$:

- The available energy is 2 880 MJ, which is equivalent to 800 KW.
- If the boilers are heated for 12 hours in a day, the energy consumption is:

  12 x 800 x 365 or 3 504 000 kW hr in one year.
- If the boiler efficiency is 70 %, energy production is:

  0.7 x 12 x 800 x 365 or 2 452 800 kW $hr.yr^{-1}$.
- The cost of heat production is thus:

$$\frac{40 \times 10^5 \text{ c}^{****}}{2\ 452\ 800 \text{ kW hr}^{-1}}$$

which is **1.63 c.unit^{-1}** or **0.45 c.MJ^{-1}** or **6.3 c.m^{-3}**

- Considering that this heat is sold, at the point of

****c = cents, 100 c = 1 South African Rand

generation, for 3.26 c.unit^{-1}***** (twice the production cost), the amount of revenue generated in one year amounts to R79 961. The pay-back period for the plant is thus less than 2 years, taking an interest rate of 20% pa. and an operating cost of R15 000 pa. into account.

(ii) **Power Generation**

The cost of electrical power from LFG has been determined by Kolbe, 1991[16], for cities with populations of 10 000, 100 000 and 1 000 000 inhabitants (Table 3). It is seen that there is a strong relationship between the number of inhabitants in a city and the cost of the electricity that can be generated from the solid waste they produce. For cities of 10 000 and 100 000 people, electricity is more expensive than that supplied by ESCOM, but for cities of 1 000 000 people, electricity generated from LFG is slightly cheaper (see Footnote*****).

TABLE 3
The Cost of Electricity Generated from LFG
(Adapted from Kolbe[16])

POPULATION	10 000	100 000	1 000 000
COST/c.kW hr^{-1}	26.4	11.6	7.0

5.3 Comparison of Different Fuel Types

Table 4 is a summary of the calorific values, efficiencies of

*****ESCOM sells electricity at 9 c.unit^{-1} to Grahamstown Municipality. The consumer pays 17 c.unit^{-1} for this electricity.

It is seen from Table 4 that the consumer cost of producing hot water with LFG is between 5 and 7 times cheaper than with coal (6.40 $c.MJ^{-1}$), electricity (5.90 $c.MJ^{-1}$), L.P.G. (5.44 $c.MJ^{-1}$) or paraffin (4.45 $c.MJ^{-1}$).

In summary, it can be concluded that for the South African situation, the use of LFG for heating purposes is financially feasible but that electricity generation is unlikely to be feasible except possibly on a very large scale.

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CHAPTER 12

LANDFILL GAS POTENTIAL IN SOUTH AFRICA

CONTENTS

1. INTRODUCTION

2. MUNICIPAL SOLID WASTE

    2.1 Incineration

    2.2 Landfill Gas

3. OTHER SOURCES

1. INTRODUCTION

It is estimated that recoverable reserves of oil (273×10^9 tonne), coal (663×10^9 tonne) and natural gas (65×10^{12} m^3) will be depleted by the year 2 150 AD (White, 1979[6], cited Rivett-Carnac[4], 1982). The possible ensuing scenarios has led to the generation of considerable interest in non-fossil fuel sources, i.e. renewable energy resources, has . Renewable energy resources include the sun, wind, water, the earth's interior and wastes.

This chapter discusses the potential for the use of municipal solid waste as a renewable energy resources in South Africa. The other resources are mentioned for sake of completeness.

2. MUNICIPAL SOLID WASTE

The use of MSW as a renewable energy source is discussed here in terms of a survey, conducted during 1990 by the National Energy Council (NEC) of South Africa[2], of MSW production in the country. The survey reveals that 3 million tonnes of MSW are landfilled annually by municipalities in the country*. Of this waste, 54.5% is domestic, indicating that the rate of domestic waste production is 224 $kg.cap^{-1}.yr^{-1}$. Industrial waste production amounts to 187 $kg.cap^{-1}.yr^{-1}$.

The survey examines the energy potential of the waste in two

---

*3 million tonne of MSW can be generated by 8-30 million people, based on a solid waste production rate of 0.1-0.4 $tonne.cap^{-1}.yr^{-1}$.

hydroelectric). In the UK, it has been estimated that renewable energy sources (largely biofuels**, wind and tidal power) can supply 18% of the energy requirements in the country. Biofuels alone are capable of contributing 33% to the renewable energy resource[3].

In South Africa, there is great potential for the use of windpumps, hydroelectric schemes and solar panels[4].

**Biofuels are produced from organic wastes such as crop residues, animal manure and degradable components of MSW.

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CHAPTER 13

CONCLUSION

The aim of this project was to investigate the subject of methane extraction from landfill with particular reference to the implementation of this technology in South Africa.

Results of experiments conducted at the Grahamstown Landfill Site in conjunction with a survey of available literature reveal that the extraction of LFG from relatively small sites (< 100 000 tonne) and sites which are in dry areas (mean annual rainfall < 500 mm); a description that applies to many South African landfills; can be feasible if the gas is used for direct heating purposes. Methods of assessing the extent of the LFG resource (kinetic modelling, flow rate measurements and pumping trials) prior to gas extraction have been described.

Further investigation of the parameters influencing LFG generation, especially moisture contents (wet vs. dry sites) and temperature, is required to facilitate (i) a better understanding of LFG generation and (ii) the development of more realistic, site-specific, models.

It has been shown that LFG extraction and utilisation need not be prohibitively expensive. In a country devoid of petroleum reserves, the utilisation of this renewable energy resource can only be sensible albeit for heating purposes as opposed to electricity generation.

APPENDIX 1

Trace Organic Compounds in Landfill Gas
(Adapted from Department of the Environment, UK[1])

COMPOUND	CHEMICAL FORMULA	OBSERVED CONCENTRATION RANGE $mg.m^{-3}$
ALKANES		
propane	C_3H_8	<0.1 - 1
butanes	C_4H_{10}	<0.1 - 90
pentanes	C_5H_{12}	2 - 105
hexanes	C_6H_{14}	1 - 628
heptanes	C_7H_{16}	4 - 1054
octanes	C_8H_{18}	9 - 675
nonanes	C_9H_{20}	31 - 226
decanes	$C_{10}H_{22}$	81 - 335
undecanes	$C_{11}H_{24}$	12 - 164
ALKENES		
butadiene	C_4H_6	<0.1 - 20
butenes	C_4H_8	<0.1 - 90
pentadienes	C_5H_8	<0.1 - 0.4
pentenes	C_5H_{10}	<0.5 - 2
hexenes	C_6H_{12}	<0.5 - 136
heptadienes	C_7H_{12}	<0.1 - 0.9
heptenes	C_7H_{14}	0.3 - 103
octenes	C_8H_{16}	< 1 - 144
nonadienes	C_9H_{16}	<0.1 - 9
nonenes	C_9H_{18}	5.2 - 75
decenes	$C_{10}H_{20}$	13 - 188
undecenes	$C_{11}H_{22}$	< 2 - 54
CYCLOALKANES		
cyclopentane	C_5H_{10}	<0.2 - 6.7
cyclohexane	C_6H_{12}	<0.5 - 103
methyl cyclopentane	C_6H_{12}	<0.1 - 79
dimethyl cyclopentanes	C_7H_{14}	0.1 - 130
ethyl cyclopentane	C_7H_{14}	<0.1 - <2
methyl cyclohexane	C_7H_{14}	1.5 - 290
trimethyl cyclopentanes	C_8H_{16}	<0.1 - 58
dimethyl cyclohexanes	C_8H_{16}	< 2 - 54
trimethyl cyclohexanes	C_9H_{18}	<0.1 - 27
propyl cyclohexanes	C_9H_{18}	<0.5 - 8
butyl cyclohexanes	$C_{10}H_{20}$	<0.1 - 4

COMPOUND	CHEMICAL FORMULA	OBSERVED CONCENTRATION RANGE $mg.m^{-3}$
CYCLOALKENES		
limonene	$C_{10}H_{16}$	2.1 - 240
other terpenes	$C_{10}H_{16}$	14.3 - 311
? menthene	$C_{10}H_{18}$	<0.1 - 29
AROMATIC HYDROCARBONS		
benzene	C_6H_6	0.4 - 114
toluene	C_7H_8	8 - >460
styrene	C_8H_8	<0.1 - 7
xylenes	C_8H_{10}	34 - 470
ethyl benzene	C_8H_{10}	17 - 330
methyl styrene	C_9H_{10}	<0.1 - 15
propyl benzenes	C_9H_{12}	36 - 292
butyl benzenes	$C_{10}H_{14}$	5.8 - 138
pentyl benzenes	$C_{11}H_{16}$	0.4 - 17.5
HALOGENATED COMPOUNDS		
chloromethane	CH_3Cl	<0.1 - 1
chlorofluoromethane	CH_2ClF	<0.1 - 10
dichloromethane	CH_2Cl_2	<0.1 - 190
chlorodifluoromethane	$CHClF_2$	<0.1 - 16
dichlorofluoromethane	$CHCl_2F$	<0.1 - 93
chloroform	$CHCl_3$	<0.1 - 0.8
dichlorodifluoromethane	CCl_2F_2	<0.1 - 48
trichlorofluoromethane	CCl_3F	<0.1 - 20
chloroethane	C_2H_5Cl	<0.1 - 46
1,1-dichloroethane	$C_2H_4Cl_2$	<0.1 - 130
1,2-dichloroethane	$C_2H_4Cl_2$	<0.1 - 8
vinyl chloride	C_2H_3Cl	<0.1 - 32
1,1,1-trichloroethane	$C_2H_3Cl_3$	<0.1 - 177
1,2-dichloroethylenes	$C_2H_2Cl_2$	<0.1 - 302
trichloroethylene	C_2HCl_3	<0.1 - 170
tetrachloroethylene	C_2Cl_4	<0.1 - 350
1,1-dichlorotetrafluoroethane	$C_2Cl_2F_4$	<0.1 - 1
1,2-dichlorotetrafluoroethane	$C_2Cl_2F_4$	<0.1 - 10
1,1,1-trichlorotrifluoroethane	$C_2Cl_3F_3$	<0.1 - 70
bromothene	C_2H_5Br	<0.1 - <2
chloropropanes	C_3H_7Cl	<0.1 - <2
dichlorobutenes	$C_4H_6Cl_2$	<0.1 - <2
chlorobenzene	C_6H_5Cl	<0.1 - 2.1
dichlorobenzenes	$C_6H_4Cl_2$	< 2 - 16

COMPOUND	CHEMICAL FORMULA	OBSERVED CONCENTRATION RANGE $mg.m^{-3}$
ORGANOSULPHUR COMPOUNDS		
carbonyl sulphide	COS	<0.1 - 1
carbon disulphide	CS_2	<0.1 - 2
methanethiol	CH_4S	<0.1 - 87
ethanethiol	C_2H_6S	<0.1 - <2
dimethyl sulphide	C_2H_6S	<0.2 - 60
dimethyl disulphide	$C_2H_6S_2$	0.1 - 40
diethyl disulphide	$C_4H_{10}S_2$	<0.1 - 0.6
butanethiols	$C_4H_{10}S$	<0.1 - 2.4
pentanethiols	$C_5H_{12}S$	<0.1 - ?1.2
ALCOHOLS		
methanol	CH_4O	<0.1 - 210
ethanol	C_2H_6O	<0.1 - >810
propan-1-ol	C_3H_8O	<0.1 - 110
propan-2-ol	C_3H_8O	<0.1 - >46
butan-1-ol	$C_4H_{10}O$	<0.1 - >19
iso-butan-1-ol	$C_4H_{10}O$	<0.1 - >5.3
butan-2-ol	$C_4H_{10}O$	<0.1 - 210
ESTERS		
ethyl acetate	$C_4H_8O_2$	<0.1 - 64
methyl butanoate	$C_5H_{10}O_2$	<0.1 - 15
ethyl propionate	$C_5H_{10}O_2$	<0.1 - 136
propyl acetate	$C_5H_{10}O_2$	<0.1 - 50
isopropyl acetate	$C_5H_{10}O_2$	<0.1 - ?6
methyl pentanoate	$C_6H_{12}O_2$	<0.1 - 22
ethyl butanoate	$C_6H_{12}O_2$	<0.1 - 350
propyl propionate	$C_6H_{12}O_2$	<0.1 - 200
butyl acetate	$C_6H_{12}O_2$	<0.1 - 60
ethyl pentanoate	$C_7H_{14}O_2$	<0.1 - 27
propyl butanoate	$C_7H_{14}O_2$	<0.1 - 100
ETHERS		
dimethyl ether	C_2H_6O	0.02 - <2
methyl ethyl ether	C_3H_8O	<0.1 - <2
diethyl ether	$C_4H_{10}O$	<0.1 - 12
dipropyl ethers	$C_6H_{14}O$	<0.1 - 220

COMPOUND	CHEMICAL FORMULA	OBSERVED CONCENTRATION RANGE mg.m^{-3}
OTHER OXYGENATED COMPOUNDS		
acetone	C_3H_6O	<0.1 - ?3.4
1,3-dioxolane	$C_3H_6O_2$	<0.1 - ?5
butan-2-one	C_4H_8O	0.4 - 38
tetrahydrofuran	C_4H_8O	<0.1 - <2
pentan-2-one	$C_5H_{10}O$	<0.1 - 4.2
methyl furans	C_5H_6O	<0.1 - 0.8
dimethyl furans	C_6H_8O	<0.1 - 12
? camphor/fenchone	$C_{10}H_{16}O$	<0.1 - ?13
carboxylic acids	$C_nH_{2n}O_2$	<0.1 - <2

REFERENCE

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APPENDIX 2

Landfill Gas Migration:
A Preliminary Investigation at the Coastal Park Landfill
Cape Town

NOVEMBER 1990

Background

The production of landfill gas (LFG) creates a positive pressure regime in a landfill site. Unless the gas is extracted, this pressure gradient will cause the gas to move convectively out of the landfill, vertically and/or laterally, according to Darcy's Law describing pressure flow. In addition to this pressure driven flow, the gas will also diffuse from the site, because of concentration gradients. Diffusive gas flow is described by Fick's First Law[1,3,5].

The phenomenon of LFG migration is cause for concern, as illustrated by a number of potentially lethal incidents involving explosions caused by LFG/air mixtures. In one such case,, a bungalow in the UK was destroyed when a spark from a central heating system ignited LFG which had migrated from a nearby landfill site and collected in the underfloor cavity of the building. One fatality related to LFG has been reported in the UK[4].

Gas will migrate along pressure and concentration gradients to the surrounding soil or atmosphere along the pathways that are most accessible and permeable. These pathways may include any installations on the landfill site, cover materials, settlement cracks, basements, floors, defective brickwork, drain pipes, soil etc[3].

The composition of LFG is likely to change as it migrates through a medium due to the different physical and chemical properties of its major constituents, methane and carbon dioxide[2].

One group of researchers[5] have found that upon migrating through low permeability soils, LFG separates into its components, with methane travelling ahead of carbon dioxide and attaining concentrations of greater than 80% as a result of the separation. This is very possibly due to the fact that the two gases have different densities. Methane (atomic weight 16) is almost 3 times less dense than carbon dioxide (atomic weight 44), and can therefore pass through low permeability materials easier than can carbon dioxide.

LFG might also take part in chemical reactions as it migrates, with the result that the absolute and relative concentrations of components may change. Oxidation reactions may occur, in which methane is oxidised to carbon dioxide, thus increasing the carbon dioxide concentration of the gas. The methane concentration of the gas may similarly increase, if dissolution of carbon dioxide in water/leachate takes place[3].

The possible chemical changes in LFG composition with migration are summarised in Figure 1[3].

Site Description

The Coastal Park Landfill is situated in Muizenburg, 500 m from the seafront, and is operated by the Cleansing Branch of the Cape Town Municipality. The geology of this young site consists of sand. A small river, 80 m from the toe of the tip, flows past on the seaward side.

Methods

As part of a separate investigation (carried out by Cape Town Municipality), three parallel series of nine boreholes were sunk at increasing distances from the toe of the tip in the direction of the river, in order to monitor leachate migration towards the river. These same leachate monitoring boreholes were used to investigate gas migration.

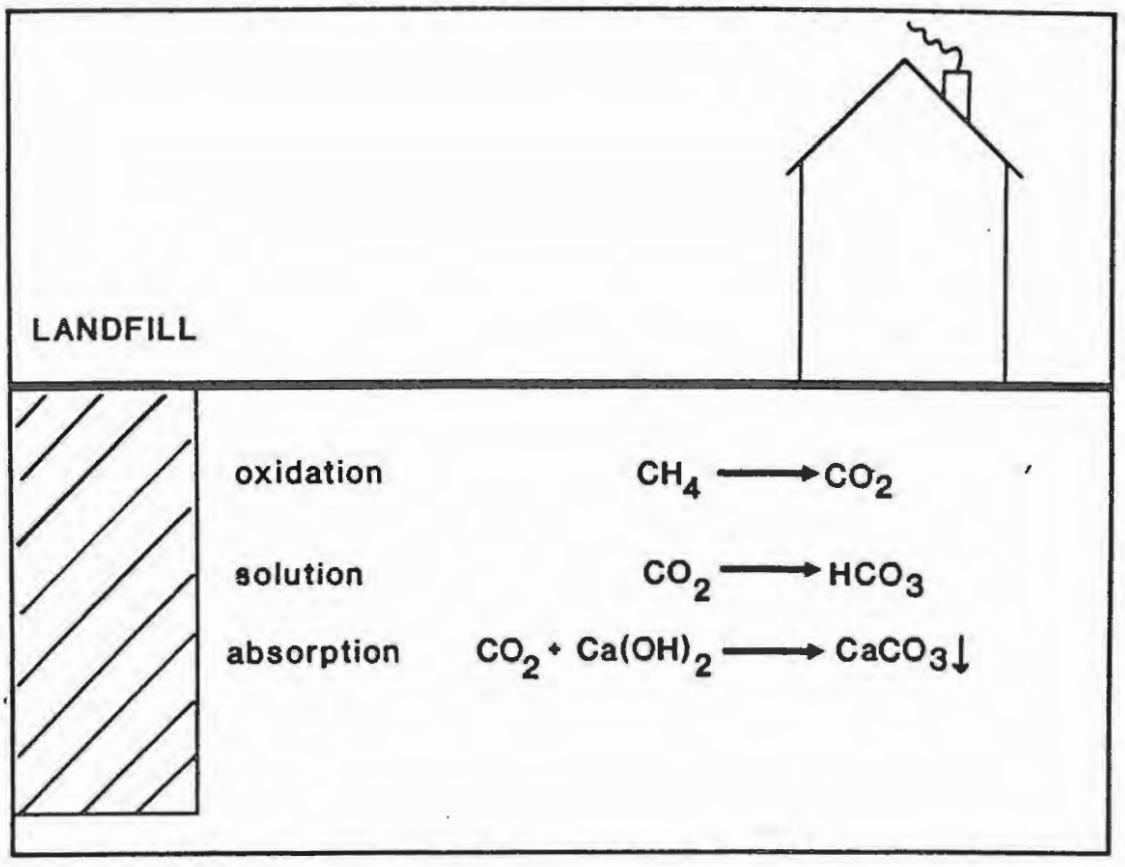

Figure 1. Chemical Changes to LFG During Migration[3]

Gas was extracted at the rate of 10 $m^3.hr^{-1}$ for 5 minutes from each well and fed directly into portable GMI gas detection instruments. The levels of methane, carbon dioxide and oxygen were measured (% by volume).

Results

The measured gas concentrations of the three series of boreholes are detailed in Tables 1, 2 and 3. Results from the three series are similar. From a graphic display of the results of series B (Figure 2), it is seen that methane and carbon dioxide appear to exhibit some stratification, with methane being detected at a distance of 2.5 m from the refuse, but no further, while carbon dioxide is detected up to 13 m from the refuse. At distances greater than 13 m, pure air is detected, 21% oxygen.

TABLE 1
Measurements in Borehole Series A

DISTANCE FROM LANDFILL m	METHANE % by volume	CARBON DIOXIDE % by volume	OXYGEN % by volume
2.5	12	20	3
5.0	0	23	1
7.5	0	15	5
12.5	0	0	21
17.5	0	1	21
22.5	0	0	21
27.5	0	0	21
37.5	0	0	21

TABLE 2
Measurements in Borehole Series B

DISTANCE FROM LANDFILL m	METHANE % by volume	CARBON DIOXIDE % by volume	OXYGEN % by volume
2.5	16	22	3
5.0	0	20	3
7.5	0	8	14
12.5	0	3	20
17.5	0	0	21
22.5	0	0	21
27.5	-	-	-
37.5	0	0	21

TABLE 3
Measurements in Borehole Series C

DISTANCE FROM LANDFILL m	METHANE % by volume	CARBON DIOXIDE % by volume	OXYGEN % by volume
2.5	10	18	12
5.0	0	4	16
7.5	0	4	20
12.5	0	0	21
17.5	-	-	-
22.5	-	-	-
27.5	0	0	21
37.5	-	-	-

Discussion

The porous nature of the sand at the Coastal Park site appears to permit some migration of LFG from the site.

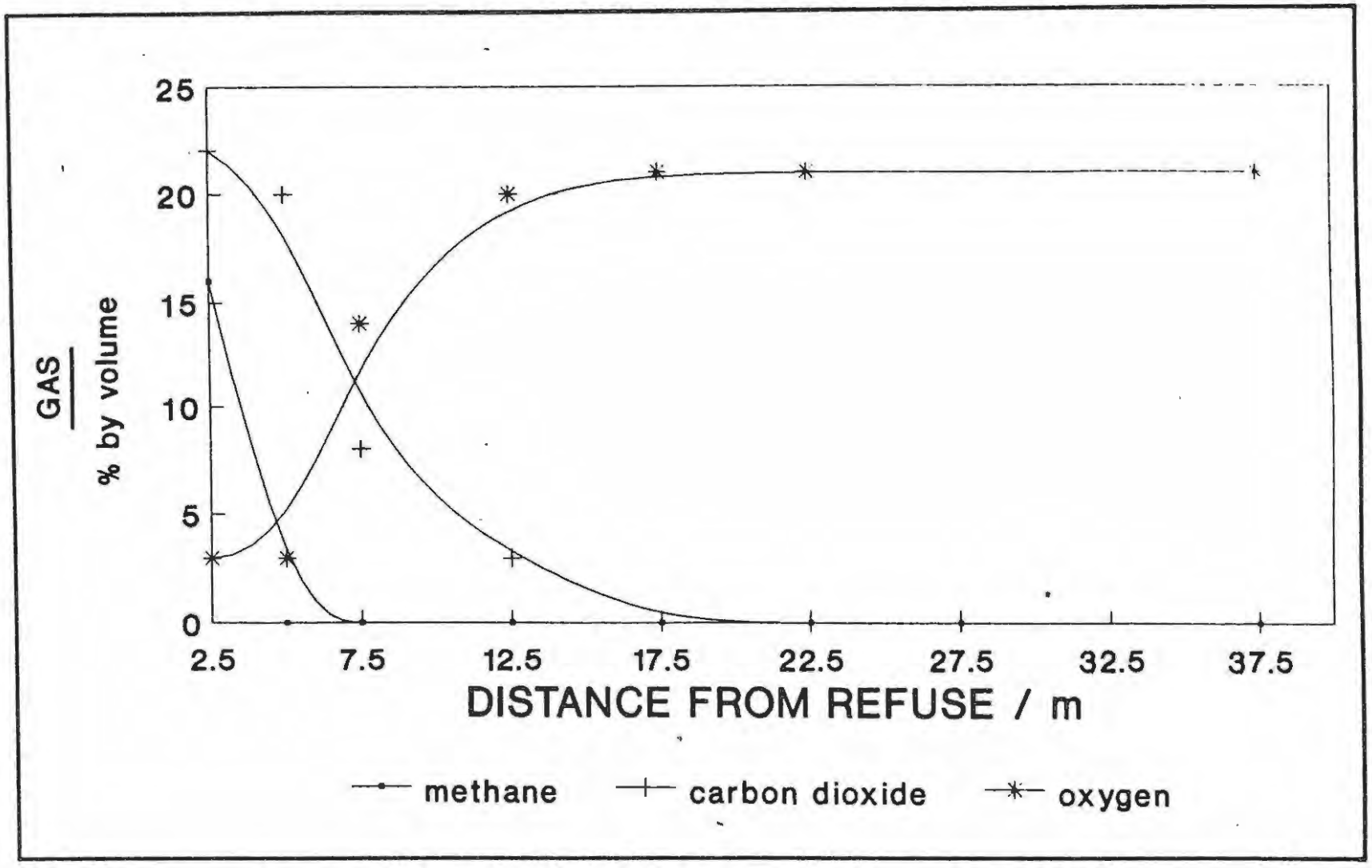

Figure 2. Methane, Carbon Dioxide and Oxygen Concentrations in Borehole Series "B" at the Coastal Park Landfill.

The total absence of methane at distances greater than 5 m from the site may indicate that the aerobic conditions which prevail in the sand result in the oxidation of methane to carbon dioxide. This oxidation may contribute to the amount of carbon dioxide. The decline in carbon dioxide concentrations is possibly due to solubility in water/leachate.

Stratification in this very porous material appears to be related to the chemical phenomenon of methane oxidation. Peterson *et al*[5] have noted stratification of the reverse type (methane moving ahead of carbon dioxide) in low permeability soil. In this case stratification can be attributed to differences in a physical property, density.

The author is of the opinion that the gas concentrations detected are not a result of LFG extraction from the landfill itself. If this were the case, methane would have been detected at greater distances from the refuse along with the carbon dioxide. It is believed that methane oxidation is not instantaneous and that were this gas pulled from the refuse, the methane would not have been oxidised by the time it reached the detector.

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5. **Peterson, E.R., Carrico, P.J. and Smithberger, J.M.** (1989). Subsurface Landfill Gas Migration: A Case History of an Air Injection System for Migration Control. Proceedings 43rd Purdue Industrial Waste Conference, Lewis Publishers Inc., Chelsea, USA. pp.221-228

APPENDIX 3

Theory of Thermocouple Psychrometry

Water Potential

Water potential is a measure of the free energy of water in a system compared with the free energy of pure water at the same temperature and pressure and is negative if the free energy is lower than that of pure water at the same temperature and pressure.
It is the sum of a number of components: matric potential, osmotic potential and pressure potential, and can be expressed in terms of pressure. Total water potential is related to relative humidity (RH)* by the following equation:

$$\psi = \frac{RT}{V_w} \ln \frac{e}{e_0} \quad \ldots\ldots (1)$$

where: ψ = total water potential (Pa)
R = gas constant (8.314 $J.mol^{-1}.K^{-1}$)
T = temperature (K)
V_w = molar volume of water (1.8×10^{-5} $m^3.mol^{-1}$)
e/e_0 = relative humidity

Equation (1) is derived from the Gibbs equation for the vapour pressure of a liquid as a function of both temperature and pressure. For a closed system with water as the liquid[2]:

$$V_w(v)\, dP = V_w(l)\, dP_t \quad \ldots\ldots (2)$$

OR

---

* Relative humidity can be defined as the ratio of the quantity of water vapour present in the atmosphere to the quantity which would saturate at the existing temperature.

$$\frac{dP}{dP_t} = \frac{V_w(l)}{V_w(v)} \quad \ldots\ldots (3)$$

where: v refers to the vapour
l refers to the liquid
P is the pressure of the vapour under the total pressure Pt
V_w is the molar volume of water

Assuming ideal behaviour, $V_w(v)$ can be substituted for RT/P:

$$\frac{d\ln P}{dP_t} = \frac{V_w(l)}{RT} \quad \ldots\ldots (4)$$

Assuming $V_w(l)$ to be constant, integration of equation (4) gives:

$$\ln \frac{P}{P_v} = \frac{V_w}{RT}(P_t - P_v) \quad \ldots\ldots (5)$$

where: P_v is the saturated vapour pressure of pure water

Rearranging:

$$P_t - P_v = \frac{RT}{V_w} \ln \frac{P}{P_v} \quad \ldots\ldots (6)$$

where: $P_t - P_v$ corresponds to water potential (ψ)
P/P_v corresponds to relative humidity (e/e_o)

A relative humidity of 1 corresponds to a water potential of zero i.e. the energy of pure water. Various relative humidity values corresponding to various water potential values are given in Table 1.

TABLE 1
Relative Humidity and corresponding Water Potentials

RH $/e/e_o$	0.999926	0.99926	0.9926	0.9296	0.48
ψ /bars	-0.1	-1.0	-10.0	-100.0	-1000.0

By measuring the relative humidity of a system one can thus obtain a value for the water potential of that system, relative humidity being linearly related to water potential in the range 0 to -70 bars**. Because water potential quantifies water status in terms of free energy, water moves from regions of algebraically higher potential to regions of algebraically lower potential.

Thermocouple Psychrometry

The Wescor psychrometer can measure relative humidity by two methods: (i) the psychrometric or wet-bulb method and (ii) the hygrometric or dew-point method. The principles of the first method, as given by Briscoe[1], will be discussed very generally, this method having been used to obtain measurements.

The construction of a soil psychrometer is illustrated in Figure 1. The small (20mm length x 5mm diameter) porous ceramic/stainless steel chamber can be buried in material such as soil or refuse and readings taken. Water vapour readily passes through the chamber so that the vapour pressure of the atmosphere inside the chamber stays at equilibrium with the vapour pressure of the material in which the psychrometer has been installed. The measurement of relative humidity is at the chromel/constantan thermocouple junction, while the copper/constantan junction is a temperature sensor. During measurement, the chromel/constantan junction is cooled to a temperature below dew point*** by an

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** 1 bar = 10^5 pascal

*** Dew point is the temperature to which a given parcel of air must be cooled at constant pressure and constant water-vapour content in order for saturation to occur. Any further cooling results in the formation of dew or frost.

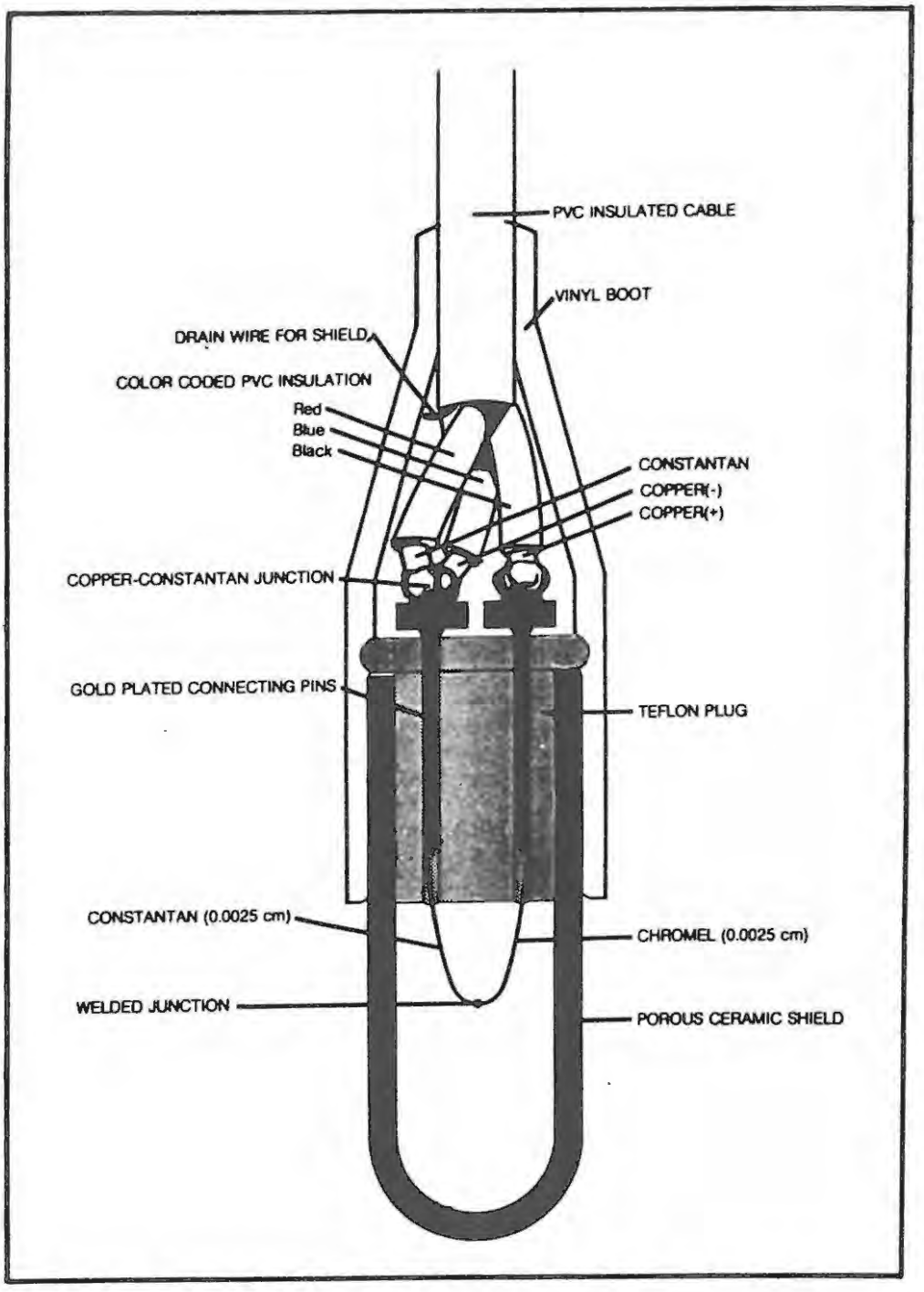

Figure 1. Thermocouple with Ceramic Shield[1].

electric current (Peltier Effect), causing water to condense on the junction. When the cooling current is discontinued, the condensed water evaporates from the junction back into the surrounding atmosphere. The heat of vaporization depresses the junction temperature, the magnitude of this temperature depression depending upon the relative humidity of the surrounding atmosphere. The drier the air, the faster the evaporation and the greater the temperature depression. The differential junction temperature, prior to cooling and during evaporation, is hence a direct function of the relative humidity and hence water potential of the system being measured. The recorder output of a thermocouple psychrometer during measurement is illustrated in Figure 2.

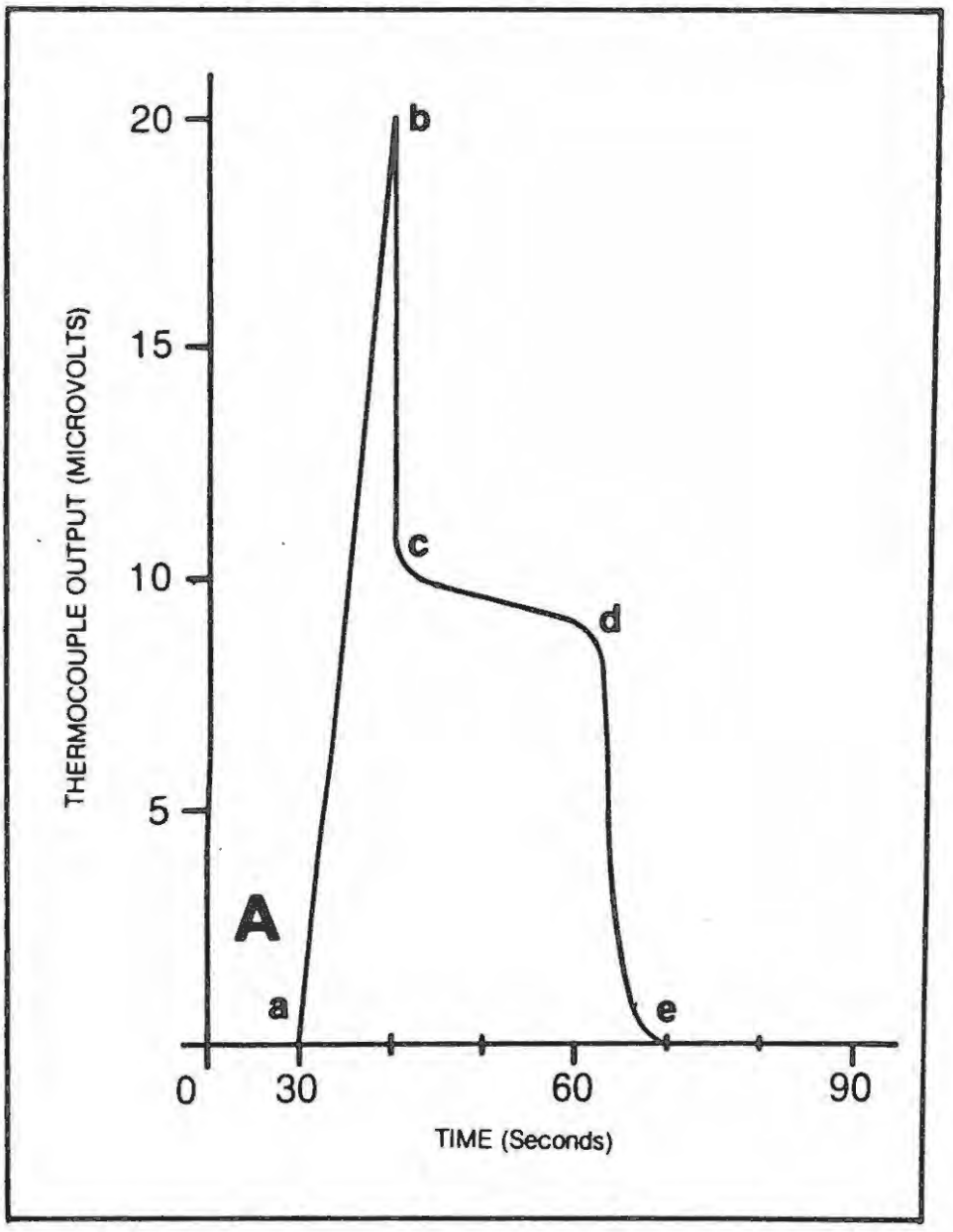

Figure 2. Recorder Output of Thermocouple Psychrometer[1].

a-b: cooling of junction by Peltier Effect
b: temperature below dew point; Peltier Current discontinued
b-c: warming of junction
c: ambient < temperature > dew point (wet bulb depression temperature)
c-d: evaporation of condensed water
d-e: warming to ambient temperature
e: ambient temperature

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APPENDIX 4

Utilisation of the Biogas Produced from the Anaerobic Digestion of Sewage: A Feasibility Study at the Grahamstown Wastewater Treatment Works

THEORY OF WASTEWATER TREATMENT

Overview

The purpose of conventional municipal wastewater treatment is to remove organic matter from solution. This is done by a combination of physical and biological processes which may or may not include the following steps[2]:

- Preliminary screening to remove large solids, and grit removal to protect mechanical equipment against abrasive wear.
- The removal of settleable organic matter, which comprises some 30%-50% of the suspended solids, and scum floating on the surface.
- Aeration in open basins and possible filtration through biological beds (trickling filters) for the microbial oxidation of organic matter, accompanied by sludge sedimentation.
- Discharge of the clear supernatant to a receiving watercourse.
- Thickening of the sludge collected from primary sedimentation
- Anaerobic digestion in order to biologically stabilise the sludge.
- Dewatering of the remaining solids either mechanically or in drying beds.
- Disposal of the dry sludge by incineration, landfilling or land application

Anaerobic Digestion

It has been stated that the sludge problem is the sewage problem[5]. This is because of the smell nuisance posed by sludge. The development of separate sludge digestion, by anaerobic biological processes, in 1910[5] was an attempt to deal with the smell nuisance of sludge and is practised at many wastewater treatment plants today. The biological decomposition of the sludge can be conducted without offensive odours and the resulting digested sludge is almost odourless.

Sludge digestion converts bulky, smelly, raw sludge to a "biologically stabilised", relatively inert material that is amenable to rapid dewatering without any offensive odours. It must be stressed that at best only 45%-50% of the organic matter originally present is removed from the raw sludge and that the remainder may resist reduction for many months[5]. Usually the pathogens in the sludge are largely destroyed, rendering the treated sludge much safer than untreated sludge for land application. Unlike pathogens, heavy metals in the sludge are not destroyed, but are concentrated, a consequence of volume reduction[5]. Despite its high nitrogen content, the sludge may not prove that useful as a soil conditioner, it being well proved by soil chemists that much of the organic nitrogen is not available under soil conditions.

The operation of an anaerobic digester, as described by Hammer, 1986[2], is as follows:

- Raw sludge is pumped from the primary settling tanks into the anaerobic digester, whereupon the contents stratify.
- A scum layer floats on top of a middle zone of supernatant (water of separation) which is underlain by actively digesting sludge and a bottom layer of digested concentrate.
- Supernatant is withdrawn from any one of a number of pipes extended through the tank wall and digested sludge

concentrate is periodically withdrawn from the digester bottom and pumped to drying beds.

- The upper half of the tank contents may be mechanically stirred and the digester possibly heated to maintain the optimum temperature of 35°C.

The microbiological process of anaerobic digestion has been discussed in detail in Chapter 2. The same basic processes occur during sludge digestion and will therefore not be mentioned here.

It is necessary that the digestion process be carefully monitored by keeping records of gas production, percentage of carbon dioxide in the gas, sludge pH and digester temperature[2]. Failure of the digester to produce biogas of a 60% methane content is more often than not the result of overloading of the digester and subsequent "pickling" of the raw sludge[2].

WASTEWATER TREATMENT IN GRAHAMSTOWN

The Treatment Plant

Waterborne sewage is screened upon arrival at the plant, to remove extraneous material, and pumped to primary sedimentation tanks. Sludge is removed daily and pumped to an anaerobic digester. Supernatant from the digester is discharged to a nearby watercourse and the stabilised sludge is pumped out to drying beds.

Operation of the Plant

An estimated population of 20 000 Grahamstown residents are served by water-borne sewage. The treatment plant receives some 1 300 m^3 of sewage per month, on average. The raw sludge collected from the sedimentation tanks is pumped to the digester at a rate of 40 m^3 per day. Between 20 and 60 m^3 of supernatant is withdrawn from the digester daily, and 1-50 m^3 of sludge every 5 to 7 days. Over a month, 1 100 m^3 of supernatant and 250 m^3 of sludge is withdrawn. The water-borne sewage entering the plant

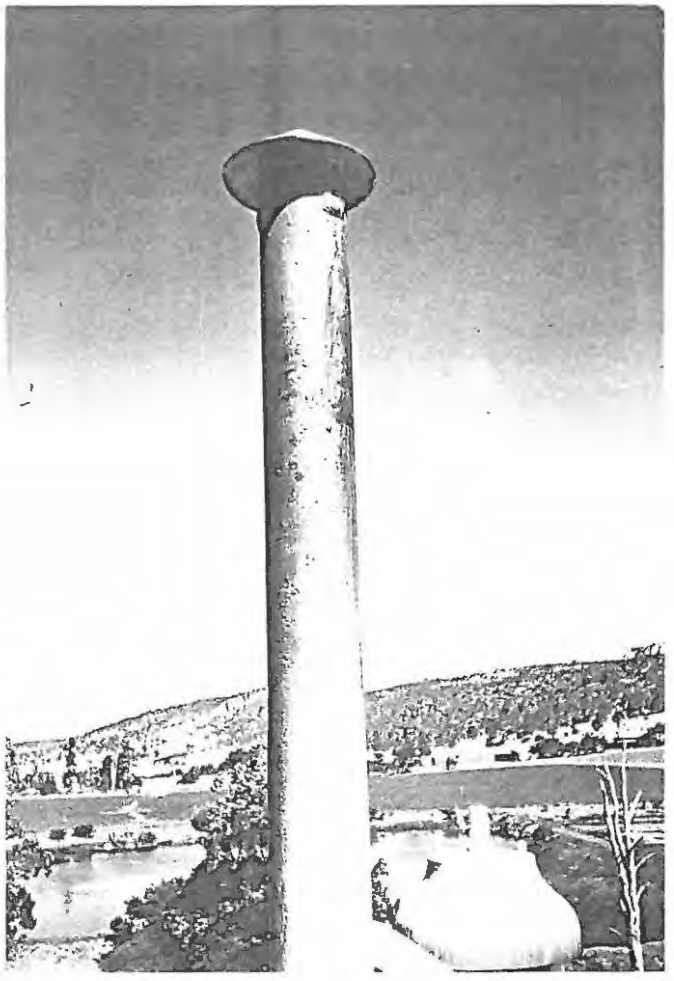
is thus ultimately converted into 99% water and 1% stable sludge. The sludge temperature in the digester, which is not heated, is low, 21 °C. It has, however, been found that upon increasing the temperature, the sludge solidifies and water has to be added to the digester, reducing its effective capacity.

The digester is not operated according to a strict schedule:

- The amount of material in the digester varies from day to day depending on the amount of sewage entering the plant daily.
- Sludge and supernatant are not withdrawn at regular intervals and the quantities withdrawn are not constant because of the above.

EXPERIMENTAL DESIGN

Monitoring at the Grahamstown Wastewater Treatment Works was carried out with the specific purpose of assessing the possibility of using the biogas produced by the digestion process as an energy source. The biogas, and hence energy, produced therefore required quantification. Measurements of gas flow rates and gas composition were taken over a number of days, at various times, on the biogas escaping from the vent at the top of the digester (Photograph 1). Composition measurements were taken by inserting a length of plastic tubing down the pipe and feeding the gas directly into portable detectors. Methane and oxygen were detected with a GMI OXYGAS-GASCOSEEKER which detects methane on a thermal conductivity basis and oxygen with the use of an electrochemical cell. Carbon dioxide with was measured with a separate GMI instrument based on infrared detection. In order to take flow measurements, a large metal tube was fitted over the gas vent pipe, with holes in the side through which the detector probe (which must be perpendicular to the direction of gas flow for accurate measurement) was inserted (Photograph 2). The instrument used was a "Testovent 4100" thermal anemometer, conventionally used in air-conditioning monitoring.



Photograph 1. The Biogas Vent at the Top of the Anaerobic Digester at the Grahamstown Wastewater Treatment Works.

Photograph 2. Monitoring Biogas Flow Rates.

RESULTS

Gas composition results are tabulated in Table 1, and flow rates in Table 2. The mode of operation of the digester at the time of measurement is also noted.

TABLE 1
Digester-Gas Composition

TIME	CH_4 %	CO_2 %	DIGESTER OPERATION
09:00	55	25	normal
10:00	56	30	normal
11:00	0	0	pumping out
12:00	44	24	stirring
13:00	46	24	stirring
14:00	50	22	pumping in
15:00	45	23	stirring

TABLE 2
Digester-Gas Flow Rates

TIME	GAS FLOW $m^3.hr^{-1}$	DIGESTER OPERATION
06:00	79	normal
09:00	71	normal
10:00	75	normal
12:00	62	stirring
13:00	84	stirring
15:00	79	stirring
16:00	84	pumping in
20:00	88	normal
22:00	71	normal
AVERAGE	77	80.4

It is readily observed that biogas composition and flow rate is not constant; there appears to be a dependence on digester

operation. During normal operation of the digester, or during stirring of the contents (usually 23 out of 24 hours), the gas contains some 50% methane and the flow rate averages approximately 77 $m^3.hr^{-1}$. While sludge is being pumped out, the methane content drops to zero. This is not surprising as it is to be expected that air is possibly drawn in through the biogas outlet when the digester is under the influence of a negative pressure.

DISCUSSION

Considering that the energy content of one cubic metre of methane is 36 MJ, and that biogas containing approximately 50 % methane is generally produced for 23 out of 24 hours at the rate of 77 $m^3.hr^{-1}$ from sludge digestion, the energy produced per hour from the Grahamstown Wastewater Treatment Works amounts to 1 400 MJ. This energy, 32 000 $MJ.day^{-1}$ (9 000 kW $hr.day^{-1}$) is not presently utilised.

The first record of the utilisation of biogas from sewage is in 1912, when gas from septic tanks was used for street lighting in England[5]. Biogas is used extensively today for the heating of sludge digesters[3] in order to maintain the optimum temperature for anaerobic digestion, 35 °C. At the Southern sewage treatment works in Durban, of the 9 500 m^3 of biogas produced daily, 2 000 m^3 is utilised for preheating sludge[6]. Biogas is also used for power generation in spark-ignition engines or gas turbines[3]. In India, a plant producing 450 m^3 of biogas per month, utilises the gas for running a 3.0 kW dual fuel generator and a pumping plant for pumping raw sewage[4]. A successful unit is also in operation in Sebokeng[3]. Some large municipal wastewater treatment works utilise the gas as a vehicle fuel, after removing the carbon dioxide[3].

The energy produced at the Grahamstown wastewater treatment works could be utilised to heat the sludge digester, which the author believes is not operating efficiently at the low sludge

temperature of 21 °C. The biogas only contains 50% methane, while most researchers give values of 60%-65% and even 70% for the methane content of biogas from sewage[3]. The inefficient operation of the digester may also be related to an inadequate retention time (5-7 days), compared to the suggested minimum retention time of 10 days[2].

The generation of electricity from the biogas is possible. Considering that the unit cost of electricity sold by ESCOM to the Grahamstown Municipality is 9 c.unit^{-1}, and assuming that (i) only 50 % of the methane from the digesters is collected for electricity generation and (ii) that the engines run at 30 % efficiency, the amount of electricity that could be generated at the treatment works in a day (1 350 kW hr) has a present day value of R120. The amount generated in a month would be worth R3 650, which is three times the electricity requirement of the plant at present (14 000 kW hr.month^{-1}, worth R1 200). The surplus electrical power could be exported to the township which is in close proximity to the plant.

Assuming that the treatment works serves a population of 20 000, energy production amounts to 1.6 MJ.capita^{-1}.day^{-1} (0.05 m^3 CH_4.capita^{-1}.day^{-1}). Values for biogas production from sewage documented by other researchers are given in Table 3.

TABLE 3
Biogas Production from Sewage

BIOGAS m^3.capita^{-1}.day^{-1}	REFERENCE
0.015	(4)
0.08	(3)
0.04	(1)
0.087	(5)

CONCLUSION

The results of this preliminary investigation indicate that the amount of energy produced by sludge digestion at the Grahamstown Wastewater Treatment Works is more than adequate for the generation of heat and/or electricity. Although the supply of this energy is not constant (biogas is only produced for 23 out of 24 hours) the gas could be stored if a constant energy demand was in fact required. The storage facility envisaged is a gasometer of the type used at the Grahamstown Landfill (see Chapter 11), which can be easily and cheaply constructed.

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