

**A COMPARITIVE STUDY OF THE DECOMPOSITION
PROCESSES AND PRODUCTS OF RICH AND POOR REFUSE IN
SOUTH AFRICA**

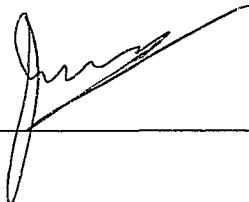
Jonathan Richard Shamrock

A dissertation submitted to the Faculty of Engineering, University of the Witwatersrand, in fulfillment of the requirements for the degree of Master of Science in Engineering

Johannesburg, 1998

DECLARATION

I declare that this dissertation is my own, unaided work. It is being submitted for the Degree of Master of Science in Engineering in the University of the Witwatersrand, Johannesburg. It has not been submitted before for any degree or examination in any other University.



28 day of May 1998

ABSTRACT

The Minimum Requirements for Waste Disposal by Landfill, published by the Department of Water Affairs and Forestry in 1994, was the first document of its kind advocating graded standards for the safe disposal of solid waste. The document introduced the concepts of affordability and technological compatibility into the ever-increasing spiral of landfilling standards. A further way to increase the affordability of waste disposal is to exploit the inherent variation of waste composition across socioeconomic levels.

In this study waste from rich and poor areas was classified and samples of the wastes allowed to degrade in large scale field cells, intermediate test cells and laboratory scale lysimeters while the decomposition processes and products were monitored.

The principal findings to date indicate that there is a large difference in the waste streams arising from rich and poor areas in South Africa. This difference does have an effect on the decomposition processes and products that occur in the landfill body, which in turn has an effect on the impact the waste has on the environment. Pollutant loads, particularly inorganic pollution, are less from poor waste. This is due to its low organic content combined with its alkali nature, which accelerates the onset of methanogenesis.

These factors can be used to further increase the affordability of landfilling standard for poor communities.

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My Parents

My wife, for her patience, help and support

DEDICATION

To Linda

Thanks for everything

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CHAPTER 1

INTRODUCTION

During the International Drinking Water and Sanitation Decade, proclaimed by the United Nations for 1980 to 1990, the main emphasis of work in developing countries was placed on the improvement of water supply and the safe disposal of human excreta. Solid waste management received comparatively little attention. As a result of the lack of research into this area, solid waste management is considered to be one of the most immediate and serious environmental problems confronting urban governments in developing countries and developing areas of countries with mixed economies (Schertenleib and Meyer, 1992). Environmental degradation and health risks in developing communities result from uncollected refuse on streets and in public areas, clogging urban drainage systems by indiscriminately dumped refuse and from contamination of water resources by uncontrolled open dumps. Inadequate collection and disposal of municipal solid waste is a major factor in the spread of gastrointestinal and parasitic diseases, primarily caused by the proliferation of insect and rodent vectors (Schertenleib and Meyer, 1992).

It could be argued that solid waste management in South Africa has been orientated to servicing the needs of a minority community in the country. Literature on solid waste issues in South Africa reflects this orientation with much emphasis on sophisticated collection equipment for formal residential suburbs, and a focus on advanced disposal systems such as sanitary landfilling. Not much attention has been paid to the issues of solid waste management in developing communities, either here in South Africa or internationally. It is only recently that the problems of waste management in developing communities started to be seriously addressed. So far the full extent of the problems are not known and even less is known on how to address them (Blight and Mbande, 1994). It was therefore considered a pressing need to start addressing these needs in South Africa. The challenge is to tackle social problems

like poverty and rapid urbanisation while at the same time protecting the environment from the effects of inadequate waste management.

Although South Africa has an industrialised sector, it cannot be regarded as a developed country. What distinguishes a developing country from a developed one? A formal definition (Campbell, 1993), is that a developing country is one where the per capita gross domestic product is lower than the average for the world. This is the most obvious way to differentiate between rich and poor countries and compares their incomes in terms of GDP or the value of all the goods and services produced in that country, adjusted for indirect taxes and subsidies (Goldthorp, 1984). A more common international variable is the gross national product or GNP which is the GDP plus any balance of income received from abroad. The problem with these indicators are numerous. They are firstly averages and conceal differences between rich and poor in the same country. Secondly, they do not consider the non-cash, traditional or subsistence sector. Thirdly, they all have to be standardised by convention to the US dollar. Exchange rates systematically underestimate the producing power of poor countries currencies and their real worth (Goldthorp, 1984). To gauge the real development status of a country we need to compare living standards in ways that do not depend on monetary value. Real indicators used to examine development status include diet, housing and sanitation, infant mortality and energy consumption. A country like South Africa can be seen to have a dual economy coupled with a disparity in development.

The dual economy is a result of a composite of two sectors, the cash. monetary or modern sector and the mainly household, traditional or subsistence sector. The disparity in development, grossly exaggerated by the political history of South Africa, results in wide disparities of worth and wealth among persons and family households in poor communities (Goldthorp, 1984). The United Nations has classified South Africa as a developed country. Notwithstanding this and the fact that communities

comprise of a range of sizes, races and socioeconomic levels, the majority of the population may be described as developing (Ball et al, 1993). Campbell (1993), describes waste management as less a matter of wealth and more one of attitude. He describes a developing country as a country where waste management practices are often basic and in some aspects inadequate for the local needs.

Solid waste management can be broken down into a number of separate phases namely:

- Generation
- On site storage
- Collection
- Transportation/Transfer
- Treatment
- Disposal

All of the above phases have historically been implemented in the country on the basis of developed countries' technologies and methodologies, with very little cognisance being taken of the appropriateness to local circumstances. This dissertation will focus on the last phase of solid waste management, namely disposal. The safe disposal of solid waste is one of the most important environmental problems which developing countries are facing today (Blight and Mbande, 1996, Ball et al, 1993, Curi, 1993). The stringent regulations which developed countries impose on the construction and operation of sanitary landfills make this solution both from the technical and economical point an unapproachable ideal for most developing countries (Curi, 1993). South Africa's problems are further compounded by the culture of non-payment for services. Non payment for services provided by Local Councils has become endemic, whilst at the same time the population growth rate for developing

communities continues at 2.4% (Fourie et al, 1997). It is estimated that approximately 15 million tons of municipal solid waste and 0.9 million tons of hazardous waste are generated annually in South Africa. All of this is finally disposed of on landfills, of which there are estimated to be some 1200 at present. Of these, standards range from burning open dumps to very well controlled sanitary landfills (Ball et al, 1993). Solid waste management activities often account for between 20 and 40% of entire municipal budget costs (Rushbrook and Finnecy, 1988). The development of rational and affordable standards for the disposal of municipal solid waste by landfilling is one way in which unnecessary expenditure on service provision may be avoided.

The science and engineering of municipal solid waste management is relatively new. Fifty years ago, most waste was disposed of in open dumps, even by authorities serving large communities (Blight and Mbande, 1994). It was only in 1959 that the first recognised definition of sanitary landfilling was published by the American Society of Civil Engineers (ASCE, 1959). At first the requirements were chiefly operational and involved the following:

1. Refuse should be covered to exclude disease generating animals and insects. This is usually achieved by covering each days deposit of refuse with a thin layer of soil.
2. Water should, as far as possible, be excluded from the refuse to minimise the formation of leachate and possible water pollution downstream of the site. This may be achieved by diverting surface or stormwater around the site to prevent it coming into contact with the waste, and by excluding rainfall from the refuse by means of compacted soil cover layers.
3. The refuse should be deposited in thin layers and compacted to reduce its volume by as much as is practically possible. Compaction also helps exclude insects and small animals, and reduces odours and smells.

Thus a definition of sanitary landfilling would be the following:

“Sanitary landfilling is an engineered method for disposing of refuse or solid wastes by spreading the wastes in thin layers, compacting each layer to the smallest practical volume and applying and compacting a layer soil to cover the waste at the end of each working day”

Most sanitary landfills in the developed world are still operated on this basis which is also the status quo for landfilling in South Africa. The design of the sanitary landfill will dictate the method of operation in a particular portion of the site, either the trench or the area method may be used.

Trench Method (See Figure 1.1) – A wide trench is dug with the sides battered back at a minimum angle of two vertical to one horizontal, so that they are stable, while the trench ends are sloped at one vertical to five horizontal to allow vehicles to enter the trench and deposit refuse. The trench should be dug slightly down contour, so that there is a slight slope on the bottom of the trench. Spoil from the trench is stockpiled along the uphill side of the trench. Refuse deposition then starts at the high end of the trench. The waste is spread out into a thin layer and compacted. At the end of the day, soil from the stockpile along the trench is used to cover the waste. When the trench is full, a second trench is dug parallel to the first and the process repeated. Stormwater from off the site is diverted around the site by means of a stormwater diversion trench. Precipitation on site is diverted around the trench by berms or banks of earth. Any rainwater falling in the trench itself will drain to the lower end from where it can be pumped, bailed or drained.

Area Method (See Figure 1.2) – An area of the site is excavated to a suitable depth so as not to encroach on the highest seasonal level of the groundwater. The excavated soil is stockpiled at the edge of the excavated area for use as cover material. Alternatively, because the trench method results in a surplus of cover soil, the area

method can be applied on top of an area previously used for the trench method. Waste is now deposited against the edge of the excavated area or against a soil or rubble berm built up of stockpiled soil, if deposition is now taking place above the original ground level. After spreading and compacting the waste, it is covered with soil. The main principle of the method is to work in an orderly and pre-planned manner and to keep the daily working face as small as possible.

As concerns and public pressure for protection of the environment grew during the 1980's, a demand arose to isolate the waste body from the environment completely by means of physical barriers – compacted clays, synthetic membranes and composite clay/membrane underliners, as well as impervious capping layers to exclude the infiltration of water precipitated on the landfill. The presence, potential benefits and dangers of landfill gas were also recognised and a demand grew for gas management of landfills. Currently, a sanitary landfill is defined as a landfill where an impermeable lining is installed and provision is made for collecting and treating leachate, managing water on and around the landfill and effectively controlling landfill gasses (See Figure 1.3) (Blight, 1994).

The majority of South Africa is a developing country, comprising of communities which cannot afford the elaborate and stringent standards prescribed by developed countries (Ball et al, 1993). South Africa has taken the first steps in addressing this problem by developing the minimum requirements for waste disposal. The next chapter will examine this document. Thereafter in Chapter 3 the landfill itself will be examined highlighting the biochemical mechanisms within it as well as its effect on the environment. In Chapter 4 the hypothesis for this dissertation is presented as well as the format for the main body of the investigation.

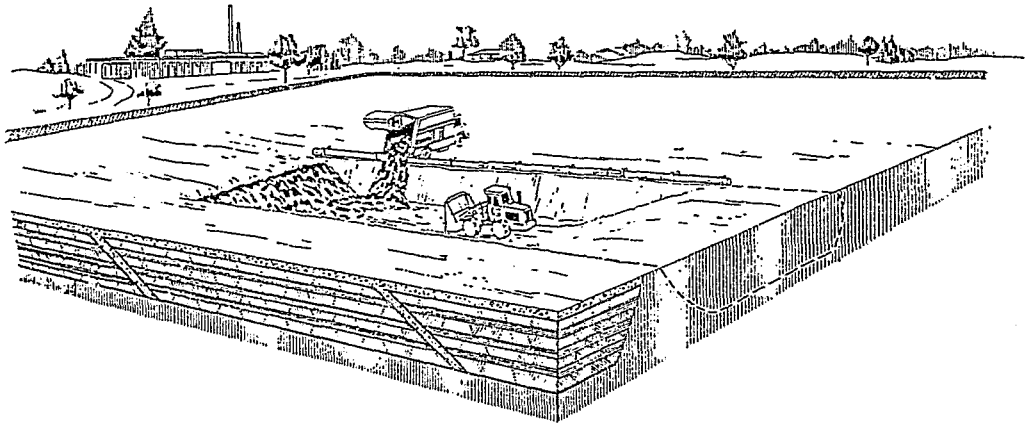


Figure 1.1 Trench method of landfilling (ASCE, 1976)

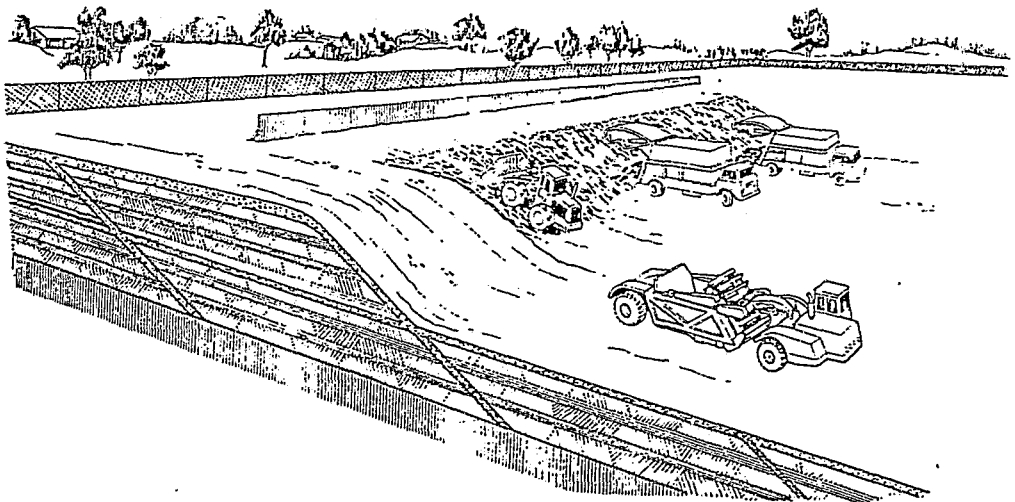


Figure 1.2 Area method of landfilling (ASCE, 1976)

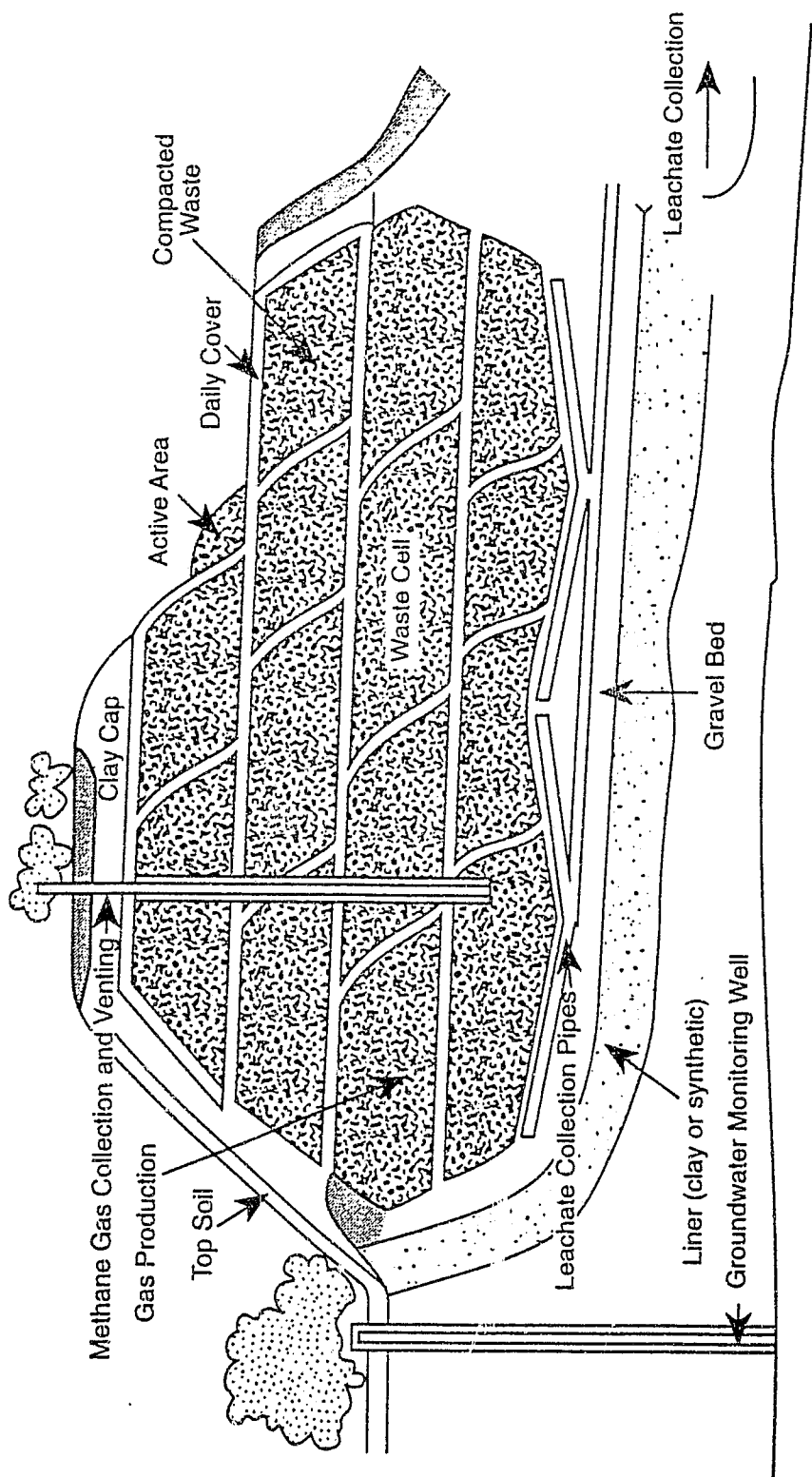


Figure 1.3 The essentials of a modern landfill (Blight, 1994)

CHAPTER 2 MINIMUM REQUIREMENTS FOR WASTE DISPOSAL BY LANDFILL

For large cities, where landfills are large and represent very appreciable sources of actual and potential ground and surface water pollution, the standards required of landfills in developing countries should ideally be on par with those required in developed countries. However, there is one qualification: the standards for landfilling in developed countries were usually formulated for the climatic condition of perennial water surplus, where every landfill produces leachate. Over much of the earth's surface, however, perennial water deficit climates prevail, and landfills may very seldom produce leachate. Under these conditions, the requirements for landfill design can be relaxed in both developing and developed countries.

For smaller communities in developing areas, having smaller landfills, and being less able to afford the costs of high standard landfilling, the requirements can be relaxed without compromising environmental protection and with the benefit of making environmentally acceptable waste disposal more affordable to the community. Possibly the first scheme of its kind for graded requirements has recently been introduced in South Africa (Blight and Mbande, 1996).

2.1 BACKGROUND

The Minimum Requirements for Waste Disposal by Landfill, were released in September 1994 by the Department of Water Affairs and Forestry (DWAF, 1994). In order to obtain affordable yet cost effective environmental protection, the BPEO or Best Practical Environmental Option approach was followed throughout the document. In other words it makes provision for environmentally acceptable yet cost-effective waste management in South Africa. The documents were introduced at an

opportune time, as the environment is under increasing pressure due to increased population and rapid urbanisation which results in growing waste generation.

The minimum requirements started off with the recognition that the majority of South Africa is a developing country, comprising of communities which cannot afford the elaborate and stringent standards prescribed in the developed world (Ball et al, 1993). Minimum Requirements are standards by means of which environmentally acceptable waste disposal practices can be differentiated from environmentally unacceptable waste disposal practices.

The general objective of the Minimum Requirements is to ensure that the most cost effective means are used to protect the environment and public health from short and long term adverse impacts of waste disposal (Ball et al, 1993). More specifically the objectives set out are:

- to improve the standard of waste disposal in South Africa;
- to provide guidelines for environmentally acceptable waste disposal for a specific landfill size and type;
- to provide a framework of minimum waste disposal standards within which to work and upon which to build.

Further to that the general objectives of environmentally acceptable landfilling are:

- to avoid the degradation of the environment in which the landfill is to be located;
- to prevent pollution of the surface and groundwater by any leachate produced;
- to ensure public acceptance by ensuring environmental acceptability.

2.2 CLASSIFICATION OF LANDFILLS

In order to achieve the set guidelines, different minimum requirements need to be appointed to different situations in a scientifically defensible way. A classification system was developed whereby landfills can be differentiated and the minimum requirements that apply to a landfill will then depend on its classification. The objectives of the classification system are:

- to consider waste disposal situations and needs in terms of combinations of inherent factors;
- to develop landfill classes which reflect the spectrum of waste disposal needs;
- to use the landfill classes as the basis for setting graded Minimum Requirements for site selection, investigation, design, operation and closure.

The Minimum Requirements use three criteria in order to classify a site for which there is a perceived need, namely waste type, the size of the waste stream and landfill operation and the potential for significant leachate generation and need for leachate management.

2.2.1 Waste Type

In the current document provision is made for two distinct types of waste namely general and hazardous wastes.

General waste (G) is a general term applied to all urban waste that is produced within the domain of local authorities. It comprises of rubble, garden, domestic, commercial and general dry industrial waste. It may also contain small quantities of

hazardous substances dispersed within it, for example, batteries, insecticides, weedkillers and medical waste discarded on domestic and commercial premises.

Hazardous waste is waste which has the potential, even in low concentrations, to have a significantly adverse effect on public health and/or the environment. This would be on account of its inherent chemical and physical characteristics, such as toxic, ignitable, corrosive, carcinogenic or other properties. The hazardous waste is further classified in terms of a hazard rating which is based on Acute Mammalian Toxicity, Ecotoxicity, environmental fate and chronic toxicity (DWAF, 1993)

2.2.2 Size of Waste Stream or Landfill operation

The ultimate physical size of a landfill will depend upon the amount of waste it receives over its lifetime. The size classification in the Minimum Requirements for general waste landfills however, addresses the size of the waste stream and the consequent size of the operation. The one characteristic that has the biggest influence on the operation of the landfill, and therefore the need for facilities, plant and operating skills, is the rate of deposition of refuse (Blight, 1994).

In order to take the time and future growth of a landfill into account and to ensure that from its inception the Minimum Requirements apply to the ultimate size of a landfill the "Maximum Rate of Deposition" (MRD) is used as the classification criterion. This figure is the projected maximum average annual rate of waste deposition, expressed in tons per day, during the expected life of the landfill.

The MRD is calculated using the following formula:

$$MRD = IRD(1 + D)^t$$

MRD = Maximum rate of deposition after t years (T/day)

IRD = Initial rate of deposition of refuse (T/day)

D = Expected annual development rate, based on expected growth in the area

t = Years since deposition started

The IRD can be calculated from one of the following sources:

- an existing weighbridge;
- existing incoming volumes converted to mass;
- per capita rates for population served.

Once the MRD is calculated, general waste landfills are divided into four size categories.

Table 2.1 Landfill size classes with respect to MRD (DWAf, 1994)

Landfill Size Class	MRD (Tonnes per day)	
Communal (C)	< 1	
Small (S)	> 1	< 25
Medium (M)	> 25	< 500
Large (L)	> 500	

Hazardous waste landfills do not take size into account and their classification are based solely on the Hazard Rating of the waste.

2.2.3 Potential for significant leachate generation and the need for leachate management.

The potential threat that a landfill poses to the environment is directly related to the possibility of that landfill producing significant leachate during its life. The Minimum Requirements acknowledges that any landfill is capable of generating leachate at some stage in its life, although this does not necessarily translate into the need for leachate management on the site. Due to the high costs involved in leachate management systems i.e. graded underliner, perhaps incorporating synthetic membranes and drains to sumps from where it is treated, a differentiation is made between sporadic leachate generation and significant leachate generation.

Sporadic generation will usually result from abnormal circumstances while significant leachate generation results mainly from climatic factors, high waste moisture content or liquid waste co-disposal practices. In order to assess in which of these categories a landfill will be, a climatic water balance is performed for the site. This is a simplified version of a fluid water balance (See 3.4 Landfill Water Balance) and uses published, easily available meteorological figures. The climatic water balance is calculated using only rainfall and evaporation data and has been found to be a conservative means of determining leachate potential as runoff and moisture storage capacity of the waste body are ignored. The climatic water balance is defined by

$$B = R - E$$

where:

B is the Climatic Water Balance in mm of water

R is the rainfall in mm of water

E is the evaporation from a soil surface in mm of water (usually adjusted A or

S pan evaporation data)

B is calculated from the wet season of the wettest year on record and then for successively drier years. If B is positive for less than one year in five no significant leachate generation is assumed to occur (B⁻). If B is positive for more than one year in five, significant leachate generation is assumed to occur and leachate management systems will be required (B⁺). High initial moisture contents of waste or co-disposal of liquid wastes will also put sites into the B⁺ classification.

2.3 APPLICATION OF LANDFILL CLASSIFICATION

Once an existing or proposed landfill has been classified according to waste type, size and climate, the minimum requirements that apply to that landfill can be identified. The minimum requirements are set out in the document in a tabular form (See Figure 2.1). For each of the following phases in a landfill's life there is a corresponding list of minimum requirements:

- site selection
- permitting
- site investigation
- environmental impact assessment
- landfill design
- site preparation
- landfill operation
- landfill operation monitoring
- rehabilitation, closure and end-use
- water quality monitoring.

The minimum requirements for site investigation are shown in Figure 2.2 as an example of how each minimum requirement is applied.

2.4 INTERNATIONAL PRECEDENT

The work done in South Africa on affordable and practical waste management standards is in sharp contrast to the rest of the developed world in which hard and fast prescriptive practices seem to dominate (ASCE, 1976, Stief, 1993).

However since the work started in South Africa, similar ideas have been espoused elsewhere in the world (Stegman, 1993). Many of the ideas expressed in this paper or "basic rules" for landfilling are similar to those in the Minimum Requirements although some of its operational aspects, like a 2m loose aerobic layer and leachate recycling will be difficult to achieve. Furthermore its prescriptive requirement for landfill gas extraction and treatment, irrespective of size of landfill, will be difficult to apply in economically disadvantaged areas

2.5 CONCLUSIONS

South Africa has taken a bold and evolutionary step in the design approach for landfills. The Minimum Requirements are a first step towards recognising not only desirability, but also affordability and practicality in waste disposal (Ball et al 1993). It is believed that this is the first set of requirements that takes direct consideration of the climatic region within which the landfill is situated. It is also believed to be the first set of requirements that specifically considers the effect of the size of the landfill on design requirements and therefore its affordability to the community it serves. As such it is ideal for the South African situation, with its diversity of socioeconomic

levels and mixture of developed and developing communities. The end goal of this work is to limit the landfills impact on the environment. The two main impacts are the gas and leachate generation which are products of the biological decomposition of the organic matter in refuse. These aspects will be dealt with in the next chapter.

WASTE TYPE	G General Waste								H Hazardous Waste	
LANDFILL SIZE	C Communal Landfill		S Small Landfill		M Medium Landfill		L Large Landfill		h Hazard Ratings 3 & 4	H Hazard Ratings 1-4
CLIMATIC WATER BALANCE	B ⁻	B ⁺	B ⁻	B ⁺	B ⁻	B ⁺	B ⁻	B ⁺		
<u>MINIMUM REQUIREMENTS</u>										
NOTES B⁻ = No significant leachate will be generated in terms of the Climatic Water Balance calculation in Section 3.4.2, so that leachate management is not required B⁺ = Significant leachate will be generated in terms of the Climatic Water Balance calculation in Section 3.4.2, so that leachate management and/or containment is required h = A containment landfill which accepts Hazardous waste with Hazard Ratings 3 and 4 H = A containment landfill which accepts all Hazardous waste, i.e., with Hazard Ratings 1, 2, 3 and 4.										

Figure 2.1 Landfill classification system (DWAF, 1994)

LEGEND B ⁻ = No significant leachate produced B ⁺ = Significant leachate produced R = Requirement N = Not a requirement F = Flag: special consideration to be given by expert or Departmental representative	CLASSIFICATION SYSTEM									
	G								H	
	General Waste								Hazardous Waste	
MINIMUM REQUIREMENTS	C		S		M		L		h	H
	Communal Landfill		Small Landfill		Medium Landfill		Large Landfill		Hazard Ratings 3 & 4	Hazard Ratings 1-4
	B ⁻	B ⁺	B ⁻	B ⁺	B ⁻	B ⁺	B ⁻	B ⁺		
Appoint Responsible Person	R	R	R	R	R	R	R	R	R	R
Classify proposed site	R	R	R	R	R	R	R	R	R	R
Eliminate areas with fatal flaws	R	R	R	R	R	R	R	R	R	R
Identify candidate landfill sites	R	R	R	R	R	R	R	R	R	R
Buffer zone	200m	200m	500m	500m	F	F	F	F	F	F
Minimum unsaturated zone	2m	2m	2m	F	F	F	F	F	F	F
Rank Sites as indicated	N	N	N	N	R	R	R	R	R	R
Site Feasibility Study	N	N	R	R	R	R	R	R	R	R
Site description	R	R	R	R	R	R	R	R	R	R
Complete permit application form	R	R	R	R	R	R	R	R	R	R
Preliminary geohydrological investigation	N	N	R	R	R	R	R	R	R	R
Preliminary Environmental Impact Assessment	N	N	R	R	R	R	R	R	R	R
Identify critical factors	N	N	R	R	R	R	R	R	R	R
Assess critical factors	N	N	R	R	R	R	R	R	R	R
Confirm no fatal flaws	R	R	R	R	R	R	R	R	R	R
Consult interested and affected parties	R	R	R	R	R	R	R	R	R	R
Confirm best site	N	N	N	N	R	R	R	R	R	R
Compile and Submit Feasibility report (including maps)	N	N	R	R	R	R	R	R	R	R
Department of Water Affairs & Forestry confirmation of feasibility	N	N	R	R	R	R	R	R	R	R

Figure 2.2 Minimum requirements for site selection (DWAf, 1994)

Since man started living in large groups, the need to dispose of his waste has become a more pressing problem. The sanitary landfill has been used as a receptor for this waste and was seen as a method of getting rid of this waste, an "out of sight, out of mind" phenomenon. In more recent years, professionals in the field are realising more and more that pollutants may leave the landfill, either carried by water or as a gas, and that these pollutants may have an impact on the environment. This has caused the emerging concept of the landfill as a bioreactor that is used to treat waste and minimise its environmental impact (Baccini, 1989). In order to understand how a landfill works as a reactor, we need to understand the decomposition processes that occur within a landfill.

3.1 WASTE DEGRADATION

The waste that enters a landfill site may have diverse origins but much of the refuse from domestic sources is putrescible (See Chapter 5 - Refuse Classification). The refuse entering a site is degraded by chemical, physical and biological processes, but by far the most predominant process controlling refuse decomposition is biological degradation. Biological degradation in turn relies mainly on the activities of bacteria.

It has been known for some time that biological processes take place in landfill sites and that these are responsible for the degradation of the organic fractions of waste (Britz, 1993). A landfill that contains substrates promoting microbial growth and at the same time offers physiochemical parameters (temp, pH, water content, etc) compatible with microbial life can be regarded as an ecosystem. Like any other ecosystem there are a diversity of organisms performing specialised functions. Microorganisms found in a landfill environment include

bacteria, algae, fungi, protozoa, viruses and helminths. Of these, bacteria are the organisms responsible for the majority of biodegradation in landfills (Britz, 1993).

3.1.1 Bacterial decomposition/digestion

When fresh waste enters a site it is subjected to biological attack by different species of bacteria that exist within the waste. Waste that is freshly deposited and compacted in a typical landfill site initially undergoes decomposition by aerobic bacteria. Aerobes require the presence of oxygen for growth, using molecular oxygen as the ultimate electron acceptor for the primary process, yielding useful chemical energy. If the site is run according to sanitary landfill practices (See Chapter 1) which involves compaction and covering of the waste, the aerobes will eventually die when the supply of available oxygen is depleted and ingress of new oxygen is prevented (Britz, 1993).

There follows an intermediate phase in which a changeover from aerobic to anaerobic conditions occurs. During this phase, facultative anaerobes are very active. These organisms are able to adjust rapidly to live and grow in an oxygen-rich or deficient environment. Relatively few organisms are obligate (do not tolerate oxygen) anaerobes and the facultative organisms ensure the anaerobic environment is maintained by consuming any oxygen present. This situation is brought about because more energy is obtained from complete aerobic oxidation than from incomplete oxidation (anaerobic reactions). These organisms therefore serve to stabilise the anaerobic environment. The remaining degradation will thus occur in the absence of oxygen or in an anaerobic environment. The process involved is called anaerobic digestion and it involves the transformation of complex organic material to methane and carbon dioxide by a wide community of bacteria (Britz, 1993).

Anaerobes are unable to grow in the presence of molecular oxygen and obtain their energy from the oxidation of complex organic matter, but utilise compounds

other than the dissolved oxygen as oxidising agents, such as carbon dioxide, sulphate and nitrate (Britz, 1993). This process can be called anaerobic digestion, the digestion being the conversion of biopolymers (proteins, polysaccharides, lipids and nucleic acids) to small units which can be assimilated and converted to carbon dioxide and methane. This process is carried out by a large (10^9 to 10^{10} bacteria per 100ml) (Britz, 1993) and complex bacterial community with specialised ecological roles and co-ordinated metabolic activity. There are at least four different trophic classes of bacteria, metabolically independent, that have a role in anaerobic digestion, the acidogens, acetogens, homoacetogens and methanogens.

The Acidogens

The acidogens are by far the largest of the trophic groups, consisting of about 90% of the landfill community (Britz, 1993). The ecological role of the bacteria is the conversion of complex organic material into simpler organic materials by hydrolytic and fermentative processes. The bacteria which can be aerobes, facultative or anaerobic utilise a large substrate range of complex organic molecules including polysaccharides, lipids, proteins and aromatics. The products of their activities, or metabolic intermediates produced, include short chain carboxylic acids (acetic, propionic, butyric), amino acids, H_2/CO_2 , one-carbon intermediates, alcohols and longer chain fatty acids.

Organic material $\Rightarrow$ acids, H_2 / CO_2 , amino acids, alcohols

The Acetogens

The acetogens are the hydrogen producing bacteria and oxidise organic acids and alcohols to produce acetate, H_2 and CO_2 . These conversions are extremely sensitive to hydrogen which must be maintained at partial pressures below 10^{-5}

atmospheres (Britz, 1993). An inhibition of this trophic group results in an accumulation of volatile fatty acids and a subsequent drop in pH.

Organic acids and alcohols $\Rightarrow$ acetate and H_2

The Homoacetogens

The homoacetogens are the hydrogen consumers that use hydrogen and carbon dioxide as well as other one-carbon molecules to produce as its only metabolic end-product, acetate. These organisms are strict anaerobes and play an important role in maintaining the low hydrogen partial pressure required. These bacteria are also able to catabolise carbohydrates, aromatic compounds, alcohols and fatty acids. Although acetate is the major end-product other metabolites such as CO_2 , ethanol, propionate, butyrate and valerate may also be formed, depending on the oxidation status of available substrates (Britz, 1993).

H_2 / CO_2 and 1 C compounds $\Rightarrow$ acetate

The Methanogens

The methanogens or methane producing bacteria are the final trophic level and the last stage in anaerobic digestion. This group of obligate anaerobes utilise H_2 / CO_2 or one of acetate, CO, formate, methylamines, methanol and always produce methane as an end product. These bacteria require a redox potential of lower than - 300 mV and a neutral pH for growth. The methanogenic bacteria are able to use a limited range of substrates and display either autotrophic, heterotrophic or an amixotrophic metabolic pattern. Substrate ranges used include acetate, formate, methanol, ethanol, mono-, di-, and trimethylamines and H_2 / CO_2 (Britz, 1993). These bacteria do not ferment carbohydrates and are therefore dependent on the other bacteria in the landfill.

H_2 / CO_2 and 1 C compounds and acetate $\Rightarrow CH_4$ and CO_2 and H_2O

It is convenient to think of the bacteria as separate trophic groups performing their individual tasks independently. However in the complex landfill environment the bacteria must interact. This is particularly true with the methanogenic bacteria which perform most of the degradation of organic matter, leading to landfill stabilisation, see Figure 3.1 (Baccini, 1989). As stated earlier most of the organic matter is comprised of biopolymers, polysaccharides, lignin, proteins, nucleic acids and lipids. These polymers cannot be transported into a cell through the membrane, and must first be hydrolysed into their oligomeric (basic) subunits. This hydrolysis can be seen as the first step in methanogenic digestion, the so called liquefaction (Baccini, 1989). The enzymes are in fact synthesized by the organisms responsible for the second step, the fermentative bacteria. These convert mono- and dimers into fermentation products mainly $C_1 - C_5$ acids and alcohols, H_2 and CO_2 . H_2 and CO_2 , methanol, formic and acetic acids can be converted directly into CO_2 and CH_4 by acetotrophic and acetoclastic methanogens. The other alcohols and acids cannot be used as substrates for methanogens and need to be converted by the sulphate reducers and obligate proton reducers.

Several sulphate reducers convert the organic fatty acids to acetate, a methanogenic substrate, but others can oxidise the substrate directly into CO_2 . This metabolism is limited by the available sulphate, and produces hydrogen sulphide.

As sulphate is scarce in waste (Baccini, 1989), the most important organisms performing such a transformation are the obligate proton reducers. The only way these bacteria can live is to have hydrogen-uptaking organisms close by (homoacetogens). The substrates will be transformed into methanisable substrates such as acetate, H_2 and CO_2 .

Hydrogen uptake will also increase the energy yield of fermentation reactions. Thus, the activity of the methanogens will favour both the obligate proton

reducers and the fermentative bacteria which will at the same time increase their energy yield and produce more directly methanisable substrates. Increasing the yield of fermentative bacteria will in turn favour the production of exoenzymes for the hydrolysis of biopolymers. These interactions are bidirectional and mutualistic so that if one of the steps is compromised it will effect the others. For these relationships to be narrow enough to increase their efficiency, several conditions must be fulfilled which will be discussed in 3.1.2.

In summary the two main processes namely aerobic and anaerobic digestion can be summarised as follows:

Aerobic bacteria

organic substrate (C,O,H,N,P,S) + nutrients (N,P, etc) + O₂ →
CO₂ + H₂O + NO₃⁻ + PO₄³⁻ + SO₄²⁻ + new cells + energy

and

Anaerobic bacteria

organic substrate (C,O,H,N,P,S) + nutrient (N,P, etc) →
CH₄ + CO₂ + N₂ + PH₃ + H₂S + new cells + energy

(Adapted from Crawford and Smith,1985)

It is useful for the understanding of the biochemical pathways to trace a few familiar compounds through the landfill bioreactor.

Proteins

Proteins metabolise in a landfill to peptides and amino acids. Amino acid deamination leads to the formation of short chain carboxylic acids, CO₂ and NH₃ (Rees, 1980). Amino acid deamination is the sole source of the branched chain,

isobutyric and isovaleric acids found in leachates. Ammonia NH_4^+ appears in high concentrations in landfills (Rees, 1980). This together with the apparent nitrogen limiting conditions which have been claimed to exist in landfill sites (Rees, 1980), is difficult to reconcile.

Lipids

Lipid digestion consists of hydrolysis of glycerides followed by the removal of two carbon units from long chain fatty acids to yield acetates (Rees, 1980).

Carbohydrates

Cellulose is the major carbohydrate found in domestic refuse. The biodegradability of domestic refuse cellulose is likely to be much greater than that of natural, untreated cellulose and the garden waste component degrades in a similar fashion to rumen contents where much of the native cellulose is still crystalline and lignified. Cellulose gives rise to glucose and cellabiose which rapidly ferments to give CO_2 , H_2 and ethanol, acetic, propionic, butyric, valeric and capraic acids.

As mentioned earlier the most important factor controlling landfill biochemistry is the physiochemical environment that exists in the landfill. The next section will deal with the various factors that influence life and growth in landfills.

3.1.2 Physiochemical environment

Crawford and Smith (1985) state that the nature of the growth and living environment of microorganisms is often defined by any of four variables: source of energy; source of carbon and other nutritional requirements; oxygen requirements and temperature requirements. To this, other authors add pH and alkalinity, redox potential, toxic compounds and xenobiotics (anthropogenic

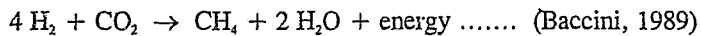
organics), water content and compaction. All of the above go to make up the physical as well as the chemical environment in which the bacteria live and the environment determines to a great extent the decomposition process that can occur.

3.1.2.1 Source of energy

Two forms of energy are externally available to organisms, light (phototrophic organisms) and exothermic chemical reactions (chemotrophic organisms). Since available sunlight plays no role in a landfill we will concentrate on chemotrophic organisms and the chemical reactions based, directly and indirectly, upon the compounds found in their environment (Baccini, 1989). This reaction will be coupled to the formation of metabolically available energy, such as pyrophosphate bonds (ATP) or trans-membrane cationic and electronic gradients. Two main types of coupling processes can be involved namely respiration and fermentation.

Respiratory processes involve an electron transport through redox intermediates, from an electron donor to an electron acceptor. If the external electron donor is an inorganic substance then the organism will be called a lithotroph and if it is an organic substance it is a organotroph.

Lithotrophic organisms use the energy and reducing power from the inorganic substrate to reduce CO_2 to synthesise new cell material. In aerobic conditions the lithotrophic metabolism is expressed when there is a starvation of organic nutrients. In anaerobic conditions, practically only H_2 can be utilised as an electron donor (Baccini, 1989). However, in the presence of nitrate acting as an electron acceptor (nitrate respiration or denitrification), reduced sulphur compounds (elemental sulphur, sulphide, thiosulphate) can be used as electron donors as well. The use of CO_2 as an electron acceptor is ecologically important as when it is coupled with the oxidation of H_2 it is the key in methanogenic digestion i.e.

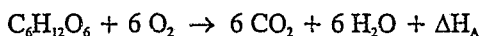


Organotrophic organisms are at the same time mostly heterotrophs as the same substrates serve in general as carbon sources and as electron donors. The same oxidants serve as electron acceptors as for the lithotrophic metabolisms, oxygen for aerobic respiration, nitrate, sulphate and elemental sulphur for anaerobic ones. Aerobic respiration gives the highest energy yield per mole of substrate consumed, but also the highest loss of energy as heat. In anaerobic digestion organotrophic methanogens convert acetate into formyl and methyl groups (respectively electron donor and acceptor) and a subsequent electron transfer then leads to the formation of methane and CO_2 .

Fermentative processes are exothermic transformations of an organic substrate, often dismutation reactions, which are coupled to the synthesis of energy rich pyrophosphate bonds. The amount of energy available from fermentation is always lower than from respiratory chains (Baccini, 1989). In most pathways leading to substrate level phosphophorylations, there is a formation of an excess of reducing power. This excess will then be transferred to a "final electron storage compound", which will then be excreted by the cell. These products (mainly alcohols and volatile fatty acids) will accumulate in the medium, unless they are consumed by other organisms. An accumulation of these products could lead to acidification of the surrounding medium.

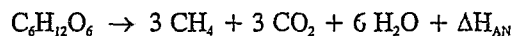
In summary if we follow the pathway of glucose decomposition with respect to generated energy (Knez, 1993) we can see the effect of the environment.

In aerobic environment



where $\Delta H_A = -2803 \text{ KJ/mol}$

In anaerobic environment



where $\Delta\text{H}_{\text{AN}} = -132 \text{ KJ/mol}$

3.1.2.2 *Source of carbon/nutrients*

Microbial biomass contains a number of elements. Some of them such as C, H, O, N, S, P are constituents of its main components, the macromolecules (proteins, nucleic acids, polysaccharides, lipids); others (metallic ions) participate in the osmotic equilibrium of the cell, in transport processes, and as cofactors in enzymatic reactions. The bacterium must find all these elements in their habitat and in turn they are used in different proportions. The first nutrient to be exhausted will act as a limiting factor for growth (Baccini, 1989). There are three major chemical elements responsible for this limitation; carbon, nitrogen and phosphorus. As can be seen in Table 3.1 these elements make up the largest proportion of the commonly occurring *E. coli* cell, the other major constituents being atmospheric gases. Although nitrogen is an atmospheric gas. it is not available to most organisms in its N_2 form.

Table 3.1 **Elemental composition of *Escherichia coli* (Rohrs, 1995)**

Element	% Dry Weight
Carbon	50
Oxygen	20
Nitrogen	14
Hydrogen	8
Phosphorus	3
Sulphur	1
Potassium	1
Sodium	1
Calcium	0.5
Magnesium	0.5
Chloride	0.5
Iron	0.2
All others	0.3

Carbon

As can be seen in Table 3.1, carbon accounts for half the dry weight of a cell. The presence of a usable carbon source is therefore a determining factor in the ability of microorganisms to grow and reproduce in a particular environment. Organisms are classified as autotrophs if they are able to use inorganic carbon (CO_2 or its derivatives). In order to reduce this source, they require an inorganic electron source. Organisms that use an organic carbon source are classified as heterotrophic organisms and they can synthesise most of their own organic material from simple organic compounds (sugars, aliphatic, aromatic, alcohols, etc) (Baccini, 1989). In addition some require already synthesised chemical structures as growth factors (vitamins, amino acids, purine, etc). Smaller molecules (monomers and dimers) can be transported directly into the cell via the semi-permeable cell membrane whereas polymers must be hydrolysed by

exoenzymes before they can be assimilated. This usually profits other members of the microbial community which are not able to produce these enzymes.

There are large differences with respect to the amount and variety of organic compounds that heterotrophs can assimilate. The methanogens are narrow specialists able to use only a few compounds, while others can use over one hundred organic compounds as sole carbon and energy source. Even organic carbon in refuse is not a homogenous fraction (Baccini, 1989). There is obviously a difference between carbon in kitchen waste and greens on one hand and in plastic or lignin on the other, so the question of bioavailability of a carbon source is also important, see Table 3.2.

Table 3.2 Carbon contents and transfer (Baccini, 1989)

Fraction	Carbon g/kg fraction	Potential g/kg refuse wwt.	Carbon %	Transfer Pot. g/kg refuse wwt
Paper/cardboard	399.7	36.2	50	18.1
Plastic	541.9	15.9	0	0
Kitchen waste	311.9	62.9	100	62.9
Fine fraction	192.3	7.8	50	3.9
Residues	-	48.2	50	24.1
Total refuse	246.9	171	63.7	109

Carbon transfer in a landfill is by far the most important elemental balance, both by gas (See 3.2) and by leachate (See 3.5).

Nitrogen

The term inorganic nitrogen includes all the major inorganic nitrogen components namely un-ionised ammonia (NH_3), reduced ammonium ions (NH_4^+), the oxyanions nitrate (NO_3^-) and nitrite (NO_2^-) and as molecular nitrogen gas (N_2).

There are a variety of intermediate gaseous oxides of nitrogen which play a smaller role in the nitrogen cycle in landfills. A simplified diagram of the nitrogen cycle is shown in Figure 3.2.

Ammonia and ammonium are reduced forms of inorganic nitrogen derived primarily from aerobic and anaerobic decomposition of organic material. The two forms exist together, either as dissolved ions or adsorbed onto other particles, and their relative proportions are a function of pH (Campbell, 1996). As cellular constituents such as amino acids are at the same oxidation and reduction level as ammonia it is the most readily utilised form of inorganic nitrogen (Rohrs, 1995). Nitrite (NO_2^-) and nitrate (NO_3^-) ions are the intermediate and end products, respectively, of the oxidation and aerobic stabilisation of organic nitrogen and ammonia. Nitrate is the more stable of the two forms and often occurs in a greater abundance.

A limited variety of organisms are able to use nitrate, but most inorganic nitrogen must be reduced to the level of ammonia before it can be assimilated (Rohrs, 1995). A lack of nitrogen in decomposing waste will lead to an arrest of physiological growth (mainly synthesis of proteins and nucleic acids) but substrate utilisation will continue for cell maintenance (Baccini, 1989). The result is an accumulation of metabolites, such as organic acids, which would decrease pH and retard certain processes, like methanogenesis.

Phosphorus

Phosphorus is often a limiting nutrient in natural systems. The most bioavailable forms of inorganic phosphorus are: orthophosphate (PO_4^{3-}), hydrogen phosphate ($\text{H}_2\text{PO}_4^{2-}$) and di-hydrogen phosphate (H_2PO_4^-) (Campbell, 1996). These forms tend to have low solubility as they occur in precipitate form. Bacteria can however utilise this form by solubilising the ions by secreting organic acids and then using

the phosphate ions in the fundamental energy units of cells, adenosine triphosphate (ATP).

Macronutrient Requirements

For a bacterial community to survive, these three basic nutrients as well as other macro and micro constituents must be available. Refuse in a landfill is generally considered nutrient deficient. An optimal ratio of carbon:nitrogen:phosphorus of 100:5:1 has been suggested (Rohrs 1995). The ratios in the *E.coli* cell in Table 3.1 is 16.7:4.7:1, the higher value of carbon in the ideal mix indicates the need for carbon as an energy source as well as for the cell structure. Literature on the composition of refuse seems to indicate that it is nutrient deficient (Rohrs, 1995) as nitrogen and phosphorus values fall below the optimum ratio. Farquhar and Rovers (1973) confirm the belief that a nutritional deficiency impedes microbial growth in landfills. Rees (1980) however, states that nitrogen and phosphorus are present in excess in landfills and do not limit microbes. The reasoning he gives stem from a classification of refuse, moisture content of 21 %, which contained 58.8% cellulose, sugar and starch, 5.7% lipids, 2.6% protein, 1.5 % plastics and 31.4 % ash, metal and glass. He thus argues:

- If 1kg refuse contains 0.47kg glucose this yields CH_4 , CO_2 and 26.3 g dry biomass. ($1\text{kg C}_6\text{H}_{12}\text{O}_6 \rightarrow 0.25\text{kg CH}_4 + 0.69\text{kg CO}_2 + 0.056\text{kg dry biomass} + 632\text{kJ heat}$ (Rees, 1980))
- If dry bacterial biomass consists of 12% N and 1.5% P (Rees, 1980) then 26.3g dry biomass contains 3.2g N and 0.4g P.
- Refuse contains 5g N and 1.9g P per kg of wet refuse, thus the required nutrients seem to be in oversupply.

He states further that N and P limitation would be expected to result in near zero concentration of these nutrients in landfill liquids, which is not the case. Rohrs (1995) indicated that nutrient supplements from the addition of sewage sludge to

refuse enhanced the degradation of the refuse. An optimum ratio of nutrients that encourages degradation was found to be 1.9 - 2.5 N : 1 P.

3.1.2.3 *Oxygen*

Aerobic bacteria require the presence of oxygen for growth, whereas oxygen is poisonous to obligate anaerobes. Deep inside the landfill one can expect to find anaerobic conditions where fermentative and methanogenic bacteria thrive. A zone will exist where oxic and anoxic layers meet at or near the top and side surface of the landfill. Knez (1993) claims this transient interface to be 2 to 4.5m in depth and Baccini (1989) claims this layer is variable, being determined by the age of the landfill. In the initial stages of degradation, high nutrient availability will cause large releases of CO₂ and CH₄, which will push the interface closer to the surface. As the landfill ages, the interface moves towards the interior of the waste mass. In this region aerobes and facultative aerobes will be found. Lastly there are the fully aerobic zones on the surface of the landfill in the region where waste has been freshly deposited.

3.1.2.4. *Temperature*

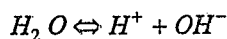
Refuse degradation can take place at temperatures between 0°C and 65°C. Within this range there are three distinct ranges corresponding to three sets of bacteria, the psychrophilic organisms (< 20° C), the mesophiles (20° C - 42° C) and the thermophiles (45° C - 65° C) (Britz, 1993). Usually maximum production rates occur at specific temperatures within each of these ranges. In microbial systems an increase in temperature leads to an increase in microbial activity. However, when the environment gets too hot more and more bacteria will become inactive or die (Britz,1993). Conversely if the temperature falls too low the microorganisms will become dormant. According to a theoretical model by Knez (1993) of the energy balance in landfills there exists three distinct temperature zones. When waste is initially placed and is undergoing aerobic degradation,

assuming ambient temperature of 20° C, the heat generated by the waste will be 69° C. When the waste is in the transient phase i.e. from aerobic to anaerobic digestion, the temperature can be up to 81° C. The anaerobic digestion process, being incomplete compared with aerobic, yields a temperature in the waste of 26° C. These temperatures are strongly influenced by external temperatures.

When the temperature of a system is changed there is a corresponding change in the microbial community structure which affects its ability to degrade waste. Bacteria are also sensitive to temperature changes within one band i.e. mesophyllic. Farquhar and Rovers (1973), state that changes of 1° C or 2° C can have a significant effect on methanogenic bacteria. Most researchers agree that the optimal temperature for methanogenic bacteria is in the range 30° C to 37° C (Farquhar and Rovers, 1973, Britz, 1993, Nozhevnikova et al, 1993, Rohrs, 1995). Rees (1980), postulates that in a biologically active site, feedback inhibition of microbial growth and metabolism by high temperatures may automatically control landfill temperature and limit its rise to the top of the mesophyllic range.

3.1.2.5. *pH and alkalinity*

The concentration of hydrogen (H^+), hydroxyl (OH^-), bicarbonate (HCO_3^-) and carbonate (CO_3^{2-}) ions is an important environmental variable that determines the ability of bacteria to survive in a particular environment. Pure water ionises slightly to produce a small proportion of hydrogen and hydroxyl ions:



(in fact the H^+ combines with a H_2O to form a hydronium ion, H_3O^+). The concentration of both H^+ and OH^- is exactly 10^{-7} mol.l⁻¹ at 24°C. The pH of a solution is defined as the negative log₁₀ of the hydrogen ion:

$$pH \approx -\log_{10} [H^+]$$

Since $[H^+]$ is 10^{-7} mol.l⁻¹ at 25°C, the pH of pure water at 25°C is 7. As $[H^+]$ increases, so pH decreases as the solution becomes more acidic, conversely as $[H^+]$ decreases, pH increases and the solution becomes more alkaline.

Carbon dioxide dissolves in water to form carbonic acid (H_2CO_3), which depending on pH dissociates to form carbonate, bicarbonate and hydrogen ions.

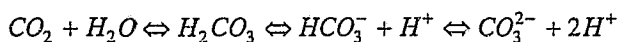


Figure 3.3 illustrates the relationship between pH and the percentage of free carbon dioxide, bicarbonate and carbonate ions in water. Note that below pH of 4, all of the CO_2 is in the form of CO_2 , above this pH the proportion of bicarbonate rises to a peak at a pH of 8.3, when carbonate ions begin to appear.

Alkalinity refers to the sum of the anions (OH^- , CO_3^{2-} , HCO_3^-) of weak acids, plus hydroxyl ions and bicarbonate ions in a sample. It is a measure of the amount of acid required to bring a sample to the inflection points in Figure 3.3. It is thus an estimate largely of the concentration of bicarbonate and carbonate ions, although other ions may contribute a small amount and is quoted as mg.l⁻¹ calcium carbonate. Alkalinity can be thought of as a measure of the buffering capacity of a system. A weak acid (or a weak base), plus its salts in water form a buffering system because the ionic equilibrium alters between the acid (or base) and its salts as the pH changes in a system in a similar manner to that shown in Figure 3.3. Thus if a solution is well buffered, and an acid is added, the pH will change less than if it were weakly buffered.

Farquhar and Rovers (1973), quote the optimal pH for biodegradation in landfills as near 7 and an alkalinity in excess of 2000 mg.l⁻¹ $CaCO_3$. Due to the synergistic

behaviour of methanogenic bacteria if the CH_4 production is inhibited, organic acids would accumulate with an accompanying decrease in pH and alkalinity. The maintenance of a near neutral pH range is due to the conversion of acidic end-products to methane by the combined activities of the acidogenic, acetogenic and methanogenic bacteria. Britz (1993) quotes the pH range for optimal digestion as 6.5-8.0. He found that the major controlling buffer in this pH range is the CO_2 , HCO_3^- and CO_3^{2-} system, with ortho-phosphoric acid, hydrosulphuric acid, the volatile acids and ammonia contributing to pH stabilisation. At lower pH values its volatile fatty acids that regulate buffering capacity.

pH has the most profound effect on the methanogens as they are reported to be neutrophilic i.e. require a neutral pH (Rees, 1980, Farquhar and Rovers, 1973). Britz (1993) however ascertains that there is evidence of acid-tolerant methane bacteria in the peat bog environment.

3.1.2.6. Moisture Content

Microbial cells consist mainly of water (75 - 90%) (Rohrs, 1995), thus a thriving microbial community requires sufficient water. Increased moisture levels in the waste mass improves the solubilisation of substrates and transport of nutrients in order to make them generally available. Water also dilutes growth inhibitors and toxins that would negatively impact on bacterial cells. Farquhar and Rovers (1973) found that moisture contents of collected refuse ranged from 5 to 50% by wet weight, far below the required 60 to 80% by wet weight and thus they postulated that in the majority of landfills the moisture content is insufficient to support maximum biodegradation.

They also found however, that excessive infiltration inhibited the activity of methanogenic bacteria. Knez (1993) in developing a model for biochemical processes in landfills found that water was produced as a result of the reactions that take place. Knez (1993) found that in the aerobic phase, 1m^3 of substrate

produced 103kg of water, 78kg in the transient phase and 8kg of water in the anaerobic phase.

Rees (1980) postulated that high initial moisture content in waste leads to high production from the fermentative bacteria. This in turn leads to a high concentration of short chain fatty acids which lower the pH and inhibit the growth of methanogenic bacteria. He believes a moisture content of 55% by wet weight is optimal for refuse degradation.

3.1.2.7. Redox Potential

In a redox reaction, electrons are transferred from one reactant to another. A substance that has a strong affinity for electrons and thus tends to attract them from other species is called an oxidising agent or an oxidant. A reducing agent, or reductant, is a reagent that readily donates electrons to another species (Skoog et al, 1988). The standard hydrogen electrode is used as a reference and the redox potential is measured against this. The sign of a standard electrode potential provides an indication of the tendency of the species involved to be reduced relative to the tendency of hydrogen ions to be reduced. A positive sign indicates that the oxidant involved is more easily reduced than hydrogen ions and that the half cell behaves as a cathode when coupled with a standard hydrogen electrode. A negative sign indicates that the species involved is a weaker oxidising agent than the hydrogen ion and that the electrode behaves as an anode. Britz (1993), states that all methanogens, being obligate anaerobes require a redox potential of lower than -300 mV. Farquhar and Rovers (1973), found that excessive infiltration stopped methanogenic activity. They postulated that the infiltration water increased the Eh of the environment. They believed that the Eh should be less than -200 mV.

3.1.2.8. *Compaction and Size of the Waste*

Compaction of waste occurs initially by compaction equipment during the placement of the waste, and then by the consolidation of the waste as successive layers are placed above it. Rees (1980) believes that compaction, along with moisture content, are the two most important factors controlling anaerobic digestion in landfills. In lysimeter experiments, with placed density of 0.2 to 0.5 t.m⁻³, no methane production was observed. To establish stable methanogenic conditions he postulated that densities of 0.5 to 1 t.m⁻³ are required. While a high moisture content and temperature promotes microbial activity, a high density tends to inhibit decomposition by making refuse carbon less available. A high density thus allows for the controlled hydrolysis and fermentation of polymers to be effectively coupled to methanogenic processes. If the fermentative stage is not slowed fatty acid accumulation occurs, This, along with high levels of CO₂, lowers the pH of the environment. An acidic environment is unsuitable for methanogenic bacteria.

Similarly if refuse is shredded or pulverised its surface area increases as does its reactivity. This increased reactivity leads to polymer hydrolysis and fermentation to fatty acids proceeding too rapidly, resulting in an inhibition of methanogenic bacteria.

3.1.2.9. *Toxic and Xenobiotic Substances*

Certain inorganic ions can inhibit microorganisms and one must remember that the anaerobic process taking place in the landfill bioreactor relies totally on a living microbial colony (Britz, 1993). This microbial colony will be destroyed by even relatively minute concentrations of poisons, aromatics, heavy metals, light metal ions, ammonia and detergents, for example chlorinated hydrocarbons, such as those used as detergents, some phenolic disinfectants and antibiotics. Organic compounds created by man that resist degradation by naturally occurring

organisms, are termed xenobiotics. These compounds if deposited in a landfill, will not degrade as the enzymes with a catalytic structure available to break them down, do not exist in nature.

3.1.3 Conclusion

In recent years many advances have been made in the understanding of the structure, biochemistry and physiology of the microbial population in landfills. We have come to see that a modern, compacted landfill is a complete ecosystem dominated by a stable, anaerobic microflora (Baccini, 1989). This knowledge helps us to understand the landfill as a bioreactor, the microbiology and biochemistry of the refuse catabolic processes, and with that understanding we can better control the effect the landfill has on the environment. The two main long term effects a landfill has on the environment is the production of gas and the production of leachate. These areas will be considered next.

3.2 GAS PRODUCTION IN LANDFILLS

3.2.1 Introduction

The production of landfill gas (LFG), in particular methane, is the result of the natural process of degradation of biodegradable wastes (See 3.1 Waste Degradation). Under normal circumstances production of LFG cannot be prevented at any site taking such wastes, but it can be controlled. The history of understanding and concern over the potential environmental impacts of LFG is relatively recent (1970s), reflecting developing awareness as a result of : changing practices, ban on waste burning, larger and deeper sites and the changing composition of incoming wastes (Petts and Eduljee, 1994). The reason LFG has been given this attention is due to the potential environmental impact it has (See 3.3 - Environmental Impacts of Landfill Gas). The section that follows will trace

broadly a generalised view of the generation, composition and characteristics of LFG.

3.2.2 Generation and Composition

LFG will be generated in any site accepting biodegradable organic matter, such as food, paper, wood, plastics, textiles, yard wastes etc. Biological and biochemical decomposition of wastes takes place over a number of years, and during this time considerable changes occur in the nature and quantity of the gas evolved. A pattern assumed to be typical and describing the sequence of events during gas production in a landfill is presented in Figure 3.4. Its development was based on the assumption that, upon placement, subsequent aeration of the refuse does not occur and that conditions within the decomposing refuse are sufficient to encourage and sustain methane generation.

The basis for this idealised curve of gas production was developed by Farquar and Rovers (1973), further modified by Rees (1980) and Augestein and Pacey in 1991. Seen together, five distinct phases of LFG production can be identified and these are designated as :

Phase 1	Aerobic;
Phase 2	Anaerobic, non-methanogenic;
Phase 3	Anaerobic, methanogenic unsteady;
Phase 4	Anaerobic, methanogenic steady;
Phase 5	Transitional atmospheric;

In Phase 1, aerobic decomposition takes place with the consumption of oxygen (O_2) present in the refuse at the time of placement. Carbon dioxide (CO_2) is produced in approximate molar equivalents to the O_2 consumed. Very little displacement of nitrogen gas (N_2) occurs. Hydrogen gas (H_2) is also produced

and the phase is highly exothermic. As oxygen is consumed anaerobic decomposition of waste commences involving several phases.

Phase 2 is the first phase in which anaerobic activity becomes dominant. During this phase fermentative bacteria are most dominant, and as a result CO_2 is the main gas produced. This period is often referred to as the CO_2 bloom stage and is often accompanied by a rise in the production of H_2 due to the acetogenic bacteria. Due to the large quantities of CO_2 being produced N_2 is displaced and the presence of atmospheric nitrogen decreases rapidly. At this stage no production of methane (CH_4) has occurred. Some researchers (Farquhar and Rovers, 1993) believe that methane production will only occur once sufficient levels of CO_2 are present in solution to act as H_2 acceptors, so lowering the hydrogen partial pressure. Once this is achieved the methanogenic bacteria become active.

Phase 3 is termed unsteady methanogenic as the percentage CH_4 is increasing, H_2 is disappearing and the levels of CO_2 and N_2 are dropping. H_2 disappears as it is a feedback for the methanogenic bacteria and N_2 and CO_2 levels decrease due to displacement by CH_4 . During this phase the gas composition tends towards a constant terminal value and this composition will continue all through phase 4.

Phase 4 is characterised by the steady production of methane rich gas, the peak quantity of methane being dependent on the prevailing conditions. Typical values of LFG composition are quoted by different authors, Following are selected examples:

Farquhar and Rovers, 1973	50% to 70% CH_4	30% to 50% CO_2
Petts and Eduljee, 1994	63.8% CH_4	33.6% CO_2
Crawford, 1985	55% CH_4	45% CO_2

As well as these main constituents, other compounds can be quantified by expensive chromatography equipment. These include alkanes, alkenes,

cycloalkanes, cycloalkenes, aromatic hydrocarbons, halogenated compounds, organosulphur compounds, alcohols, esters and ethers (Waste Management Paper No 26, 1986).

Phase 5 occurs once the biodegradable substrate is exhausted and some form of nutrient deficiency sets in. The methanogenic bacteria cease production and the gas composition begins to return to that of the natural soil conditions.

The time it takes to complete the above phases varies according to numerous factors. Farquhar and Rovers (1973) quoted Ramaswamy (1970) for the completion of phases 1, 2 and 3 of 180 days but their data showed 250 days. They also quoted Beleuche (1968) who set the time period at 500 days. All of the above data was for refuse degradation in lysimeters and the results are atypical of actual landfill conditions where all five phases can be occurring at the same time. It is important to realise that Figure 3.4 must be seen as a schematic representation of the chain of events that occur when refuse degrades. Confusion seems to have arisen in the literature when work done by Pohland et al (1983) was misquoted and actual time values assigned to the phases described above (Rohrs, 1995). Augstein and Pacey (1991) give the following time frames for the degradation process:

Phase 1 - hours to weeks;

Phase 2 - 1 to 6 months;

Phase 3 - 3 months to 3 years;

Phase 4 - 8 to 40 years and

Phase 5 - 1 to 40+ years.

3.2.3 Gas Yields

Landfill gas generation rates and gas yields per ton of substrate are a point of much speculation in the available literature. Most published gas yield data is from

lysimeter studies which are difficult to extrapolate to a full scale landfill. Crawford and Smith (1985) quotes landfillgas (LFG) yields of 30 to 180 m³ per ton of dry refuse. He then goes on to calculate the theoretical yield of LFG from refuse using the following assumptions:

- 50% of the dry weight of refuse is present as organic carbon.
- 90% of this organic carbon is converted to gaseous end products.
- Gas composition is 55% CH₄, 45% CO₂.

If 1 mole of carbon = 12g

1 mole of gas occupies 22.4 l at STP (20°C & 1 atm),

then 1 mole organic C yields 0.9 x 0.55 moles CH₄

$$1 \text{ g organic C yields } 0.9 \times 0.55 \times 22.4 / 12 \text{ l of CH}_4 \\ = 0.94 \text{ l CH}_4$$

And since 50% dry weight of refuse is organic C,

1 g of refuse will yield 0.924 / 2 l of CH₄

or 1 tonne of dry refuse yields 460 m³ of CH₄.

As an explanation of why yields in the field are different from laboratory he sites the following:

- much of the free organic C is washed out as leachate,
- conditions for bacterial growth are far from optimum.

Rees (1980) gets a theoretical gas yield by examining the yield from glucose:

1 kg C₆H₁₂O₆ → 0.25 kg CH₄ + 0.69 kg CO₂ + 0.056 kg dry biomass + 632 kJ heat.

This is equivalent to 0.7 m³ gas per kg of glucose at STP. Since the available organic C in refuse is present in carbohydrates, fats and proteins, he predicts the gas yield will be lower, about 400 m³ per tonne. Ehrig (1991) found gas yields of 120m³ per tonne at STP and he also found that the actual gas production rate decreases in an exponential form, with a half life of 4 to 7.5 years. This decay curve was also found by Crawford and Smith (1985). Baccini (1989) found gas production to occur in three main phases,

1. Large bloom with the initial aerobic phase,
2. Quiet phase during the unsteady period
3. Active production during full methanogenesis.

3.2.4 Factors affecting gas yields

Any environmental influence which impacts on the bacteria in a landfill will impact on gas production rates (See Section 3.1.2 - Physiochemical environment). Other factors related specifically to gas yields include waste composition i.e. available carbon, landfill design and landfill operations.

3.2.5 Conclusions

When landfill gas is analysed, either from lysimeters or full scale landfills, the gas data provides a useful tool to gauge at which stage in the decomposition process the waste is. It therefore provides a window in the black box of landfill biochemistry. If landfill gas is to be extracted and utilised either as an energy source or as a chemical feed stock then careful attention must be paid to the effect that landfill design and operational decisions will have on final gas yields. However if, like most developing countries, the gas is simply left to vent off into the atmosphere, then the environmental impact of these gases must be considered.

3.3 ENVIRONMENTAL IMPACT OF LANDFILL GAS

As discussed in the section above, LFG is produced in five distinct phases. Each of these phases produces different mixtures of gasses and these gases have an effect on the environment in which the landfill is situated. Of all of the phases of gas production, by far the most prolific and longest lasting is the methanogenic phase. In this phase the two main gasses produced are methane and carbon dioxide.

3.3.1 Methane

Methane is odourless and colourless and only slightly soluble in water. Methane is a flammable gas and when mixed with air within certain limits, termed the lower explosive limit and the upper explosive limit, may ignite to produce fire and explosions in open and confined spaces respectively (Petts and Eduljee, 1994). The flammable limits of methane are 5 to 15% by volume in air. The presence of carbon dioxide affects these ranges but there is little change at the lower limit. Methane mixtures in which oxygen is deficient, while not igniting at source, may present a potential risk at the periphery due to dilution with air. Poor mixing may enable flammable volumes of gas to exist even where the overall composition may appear to be outside of the explosive limits.

Methane is approximately 30 times more effective as a greenhouse gas than carbon dioxide (Petts and Eduljee, 1994). Landfills are now considered to be the largest source of methane emissions in the UK (29%), followed by cattle (23%) and deep mined coal (22%) (Macdonald and Fleming, 1994).

3.3.2 Carbon Dioxide

Carbon dioxide is a non-combustible, colourless and odourless gas which is highly soluble in water. In confined spaces CO₂ presents an asphyxiation risk (Petts and

Eduljee, 1994). Although not strictly toxic, it has physiological effects which are not simply related to the displacement of oxygen. Elevated levels of CO₂ affect respiration and can prove fatal. An exposure to a concentration above 5% by volume represents an acute danger to life. When CO₂ dissolves in water it lowers the water's pH by forming carbonic acid (H₂CO₃) (See 3.1.2.5). Carbon Dioxide is considered to be the main component of greenhouse gas on earth.

3.3.3 Other Constituents

A number of other gasses occur at different stages in the landfill's evolution. These are briefly summarised below.

- Hydrogen - Hydrogen is produced by acetogenic bacteria and hydrogen production in phases 2 and 3 can peak at 20% of LFG (Waste management paper no. 26, 1986). Hydrogen is a flammable gas when mixed with air and has a lower explosive limit and upper explosive limit of 4 to 74% by volume in air.
- Hydrogen sulphide - If high concentrations of sulphate bearing wastes are present e.g. plaster board, then sulphate producing bacteria produce hydrogen sulphide. This gas is a colourless, odourless, toxic gas which is flammable (Explosive range of 4.4 to 45% by volume in air (Petts and Eduljee, 1994).
- Minor constituents - Some of the minor constituents of LFG are considered toxic, carcinogenic, mutagenic and tetragenic. The trace compounds in domestic waste present more of an odour nuisance problem owing to the presence of thiols, esters and sulphides.

3.3.4 Landfill Gas Migration

Gasses will flow from areas of high pressure to areas of low pressure. The underlying mechanisms responsible for the migration of LFG are provided by the

pressure developed within the site, the pressure gradient within the surrounding strata and the concentration gradients of the component gasses (Petts and Eduljee, 1994). The extent of gas movement in and beyond a site boundary is governed by these mechanisms and the relative permeabilities of the waste and surrounding strata, see Figure 3.5. Rising atmospheric pressure will reduce the tendency for gasses to diffuse upwards through the soil, whereas falling pressure will encourage release of gasses.

Migration pathways in the surrounding ground may be through permeable strata such as sand and gravel, or open, natural or artificial pathways such as fractures and fissures, mine workings, service ducts, sewers, land drains, street lighting cables etc. LFG will change in composition as it migrates through the soil and strata by oxidation and dissolution mechanisms, which means that it will not necessarily be the same as that measured at the site during a monitoring program.

3.3.5 Impacts of Landfill Gas

Landfill gas represents as great a risk during the operational phase as during the post closure phase, and the impacts will continue till all the available biodegradable material is consumed. A summarised version of potential impacts are listed below (Petts and Eduljee, 1994, Waste management paper no. 26, 1986):

- Explosions due to methane migration and accumulation in confined spaces.
- Flash fire in an open space with the potential for the waste in the fill to be ignited if release of the gas is through a fissure to the surface.
- Asphyxiation of people entering culverts, trenches or manholes on landfill sites.
- Vegetation and crop stress and loss due primarily to displacement of soil oxygen in the plant root zone and subsequent anoxic conditions.

- Some trace compounds may also be phytotoxic. Damage symptoms resemble those of a drought.
- Nuisance effects due to the odourless trace compounds.
- Water pollution, in particular CO_2 which is highly water soluble and will increase water hardness producing an acidic solution.
- Health effects from emissions of trace compounds.
- Contribution to greenhouse gasses.

3.3.6 Gas Control Measures

Three types of systems may be used, either singly or in combination, to prevent the migration of LFG. These are impermeable barriers, passive venting and gas pumping.

Low-permeability barriers such as synthetic liners, will impede gas flow more efficiently than natural soils. Synthetic liners, while providing much lower permeability, do nevertheless allow gas to pass through and they are usually used in conjunction with other systems including clay, bentonite and PFA/cement grouts.

Passive control systems act by surrounding the site with a zone of high permeability material through which the gas can readily vent to the atmosphere. They usually consist of either gravel filled perimeter trenches, often with a membrane on the outer wall or vent wells going directly into the waste. Care must be taken to prevent sealing of the top surface of the vents. Venting directly to atmosphere will create odour and nuisance problems if it is too localised.

Active control systems prevent lateral migration by extraction of the gas using vacuum pumps through a number of vent wells installed directly into the waste. Usually perforated pipes are inserted into the wells which are then backfilled with gravel. The abstracted gas is taken to flare stations or steam turbines and burnt to

CO₂ before being released into the atmosphere. Care must be taken not to overextract. This would introduce oxygen into the waste which is toxic to the methanogenic bacteria.

3.3.7 Post methanogenic phase

In a normal compacted landfill, the methanogenic phase will probably last 10 to 30 years (Baccini, 1989). After this time, the landfill has become a relatively inert soil-like mass, whose gas production should no longer require control. Very limited work has however been done on this phase of the landfill's evolution. Baccini, 1989 gives a theoretical chain of events:

- At the interface between oxic and anoxic layers at the surface and side of the landfill, the rate of production of substrates that can be rapidly oxidised aerobically decreases.
- The oxic-anoxic interface will then move towards the interior of the waste mass, when the speed of oxygen diffusion exceeds its consumption rate.
- The humigenic materials, such as lignin, which were not decomposed anaerobically, will then undergo a slow transformation into humic substances by mineralisation.
- Certain reduced minerals (ammonia, sulphur or sulphides) which could have accumulated in anaerobic conditions could be biologically oxidised by aerobic chemolithoautotrophs, leading to the formation of nitric and sulphuric acids. To what extent this could bring further leaching of metals depends on the buffering capacity of the waste mass.
- The waste mass could then behave like an organic peat soil undergoing slow mineralisation. This mineralisation could be accompanied by a release of minerals into the leachate.

3.3.8 Conclusions

Landfill gas is but one of the major effects a landfill has on the environment in which it exists. The other major effect is that of deteriorating water quality due to the release of leachate into the environment.

3.4 LANDFILL WATER BALANCE

3.4.1 Introduction

For a landfill to impact on the environment it is necessary for pollutants to leave the landfill. The two main routes by which pollutants leave a landfill are by gaseous emissions (See 3.3 Environmental impact of landfill gas) and leachate. Leachate is generated as a result of the percolation of water or other liquids through any waste and by the compression of the waste due to its weight (Parsons, 1994). This leachate can be defined as a liquid that is formed when water or another liquid comes into contact with decomposing refuse. It is a highly contaminated aqueous solution which carries in it dissolved solids and the final and intermediate products of waste decomposition. The constituents have the potential to migrate to both groundwater and surface waters, with associated contamination of drinking water resources and deleterious effects on aquatic ecosystems.

The water balance or water budget technique has been developed with the aim of allowing the pollution potential of a landfill to be assessed from a knowledge of climatic and hydrological parameters and with some assumptions regarding the properties of a landfill (Blight et al, 1992). The method is based on the principle of conservation of mass and assumes that the landfill system is closed. The mass of water entering the system must equal the mass of water leaving the system + mass of water retained in the system (Parsons, 1994).

i.e. leachate generation = water input - water losses $\pm$ change in storage

If individual parameters are regarded, then the equation can be expressed as:

$$L = (P + W_m + B + L_i) - (R + E_t + G) \pm \Delta S$$

where L = leachate generation

P = precipitation

W_m = initial moisture content of the waste

B = chemical and biological water production

L_i = lateral inflow

R = runoff

E_t = evapotranspiration

G = vapour loss in gas

ΔS = change in storage

In the following sections each of these elements will be considered in turn and their relative effect on the water balance equation evaluated.

3.4.2 Precipitation

Precipitation in its general sense includes rain, hail, sleet and snow. In a South African context however, the first two are of general significance. Precipitation is the basic source of water input into a landfill system and is also the element in the water balance equation which is the easiest to measure, using a standard rain gauge (Hojem, 1988). It is important to realise however that only the portion of precipitation that infiltrates the cover will change the moisture balance in a landfill. The water input into landfills by precipitation has also been found to be dependent on the nature and duration of rain events as well as the standard of operations on the site. The size of uncapped working area of landfill surface is of

particular importance due to high permeabilities, as well as areas where rainfall can pond and infiltrate. High intensity, short duration rain events will be of greater significance in impacting working fills, whereas for closed fills which have a cap or surface seal, it is the long duration, low intensity event which could produce the greatest potential overall infiltration through the capped surface (Stone, 1996).

3.4.3 Initial Moisture Content of Waste

The amount of water held within the waste when it is deposited can also be seen as the initial moisture storage in the waste. Generally the component of the waste stream which is readily biodegradable i.e. vegetable and putrescible and paper, is the component which holds the majority of the initial moisture. This amount of water is determined by oven drying a sample and is usually expressed as the ratio of the mass of water to the mass of solids i.e. $w = M_w/M_s$ (Craig, 1987). This is the definition derived from classical, geotechnical engineering and is referred to as the moisture content by dry weight.

Hojem (1988) when simplifying his water balance equation stated that this is one of the components usually considered to be measurable. In tests he did however, he assumed an initial moisture content of waste to be 20% dry weight. Rohrs (1995) who worked on codisposal of refuse with sewage sludge found that the initial moisture content of sludge could raise the initial moisture content of waste from 30% wet weight to 40% wet weight.

3.4.4 Chemical and Biological Water Production

This element of the water balance equation is usually assumed to be negligible (Hojem, 1988). Knez (1993), set up a biochemical model of landfill behaviour in which he found that in all anaerobic phases for a 1 m³ element within the landfill that there was a net gain of 78 kg of water.

3.4.5 Lateral Inflow

Lateral groundwater inflow could be caused by extreme rainfall events or prevailing geohydrological conditions e.g. perched water table. This is one element in the equation totally in the hands of the design engineer and it should not even be considered if proper site identification and selection have taken place.

3.4.6 Runoff

Runoff is that part of rainfall that does not infiltrate the earth's surface, nor is it lost through evapotranspiration. Runoff is one of the few factors that can easily be controlled with good site management. With proper design and operation, runoff can be prevented from entering the site while runoff from site can be promoted using a low permeability sloped cover (Parsons, 1994).

Runoff can occur by two mechanisms, namely overland flow and interflow. Overland flow occurs when rainfall intensity exceeds infiltration capacity while interflow is that flow which cannot infiltrate vertically and thus flows horizontally at or near a subsurface in the vadoze zone. The rate of runoff is affected by the characteristics of the catchment area i.e. antecedent soil moisture conditions, permeability and infiltration capacity of the soil, surface slope and the amount and type of vegetation cover. Runoff is also affected by the intensity and duration of rainfall events.

3.4.7 Evapotranspiration

Evapotranspiration is the term used to include both evaporation and transpiration. Evaporation is the changing of water from a liquid to a gaseous phase whereas transpiration is the term given to the passage of water from the ground through the roots of plants into the leaves and then to atmosphere. It is practically impossible to separate these two terms in the field and hence the term

evapotranspiration is used. It can however be noted that evaporation plays a bigger role on waste sites due to the sparse vegetation cover in the operational phases. In the post-closure phase however, transpiration will be more important. Evapotranspiration is arguably the most difficult term in the water balance equation to measure with confidence (Parsons, 1994). It is impacted upon by temperature, wind, atmospheric pressure, daylight hours, sunshine hours, topography, solar radiation, albedo (reflective ability of a surface), dissolved solids, vegetation, the nature of the surface and relative humidity.

The only way to make a direct measurement of this parameter is by an evaporation pan. The figure obtained from this is then used to calculate, with an empirical correction factor, potential evapotranspiration. If a A pan is used the figure is multiplied by 0.7, if a S pan is used the figure is multiplied by 0.9 (Blight, 1994). Recently the Bowdens ratio apparatus has been developed which allows for the direct measurement of evapotranspiration. For on-site measurement however the robustness and simplicity of the evaporation pan is preferred.

3.4.8 Vapour Loss in Gas

During biochemical breakdown, water vapour is lost as one of the components in landfill gas. Knox (1991) quoted by Parsons (1994) said that a maximum of 10 liters per tonne could be lost.

3.4.9 Change in Storage

The moisture storage capacity, or field capacity of a material, is that water content that a porous material (refuse or soil cover) will absorb and store within its pores by capillarity (Blight et al, 1992). If a quantity of water is added to a material it will absorb it with a subsequent change in storage. If a quantity of water is added to a material already at its storage capacity, an equal quantity will drain out of it to restore moisture equilibrium. The storage capacity of waste is a simple concept

as stated above but when viewed in practical terms, a number of complexities arise.

- Heterogeneity - Waste is heterogeneous and compacted waste in a landfill will vary from place to place, with each segment, depending on its constituents, having its own moisture storage capacity. Parsons (1994) presented a table by Knox (1991) to illustrate this point, see Table 3.3.

Table 3.3 Absorptive capacities of paper based refuse components (Parsons, 1994)

Material	Absorptive Capacities (% dry wt)
Glossy Magazine	100
Solid and corrugated cardboard	160
Newsprint	300
Paper towel	450
Toilet Tissue	700

- Preferential flow - Preferential flow paths, also referred to as channeling and short circuiting, exist within the waste body. These result in leachate being detected before actual field capacity has been reached.
- Temporal changes - An aspect of moisture storage that needs to be considered and which is probably unique to the waste management field is that of how properties change with time (Parsons, 1994). This is due to factors like decomposition of waste, settlement induced by the weight of the waste and subsequent densification of the waste mass leading to a decrease in the storage capacity.

All these factors lead to estimates for field capacity of waste varying dramatically between authors. Crawford (1985) quotes ranges of 50 to 60 % dry weight although he uses theoretical values in a range of 40 to 133 % dry weight.

3.4.10 Conclusions

The net consequence of a landfill water balance is that as the amount of water entering the system increases the likelihood of leachate being produced increases. Good landfill operations management can help to retard or even stop the production of leachate, especially in semi-arid and arid regions.

3.5 LANDFILL LEACHATES

3.5.1 Introduction

Leachate can be defined as a liquid that is formed when water or another liquid comes into contact with decomposing waste. As the percolating water moves through the waste it deteriorates in quality resulting in a liquid which may vary in colour from light brown to nearly black, usually has a sweetish and sickly smell, and often exhibits an iridescent sheen on the surface. Its pollution potential can be 10 to 100 times that of raw sewage (Petts and Eduljee, 1994).

The main components of landfill leachate can be summarised as follows:

- Major ions - calcium, magnesium, potassium, iron, sodium, ammonium, bicarbonate, sulphate and chloride
- Trace metals - manganese, zinc, copper, chromium, nickel, lead and cadmium.

- A wide variety of organic compounds usually measured as total organic carbon (TOC), chemical oxygen demand (COD) and biological oxygen demand (BOD).
- Microbiological components

Leachate quality is primarily dependent on the balance between the acetogenic and methanogenic phases of refuse degradation. It is however, subject to processes within a dynamic environment which act to change the composition of the liquid by diffusion, solution and precipitation of the components, so that the composition cannot be predicted accurately. The values in the literature vary greatly with the result that what follows is a summarised version of the typical ranges expressed by the authors.

3.5.2 Acetogenic Leachate

Leachate produced in the fermentation/hydrolysis phase of putrescible breakdown is characterised by a high organic strength and low pH (Gettigny et al, 1996; Waste management paper no. 26, 1986; Ehrig and Scheelhaase, 1993; Petts and Eduljee, 1994; Crawford, 1985). Robinson and Gronow (1993) define acetogenic leachate as having concentrations of volatile fatty acids greater than 1000 mg/l and chloride values greater than 500 mg/l. COD ranges from 20 000 to 40 000 mg/l. Ehrig (1991) used the ratios of $BOD_5:COD$ greater than 0.4, low pH and significant concentrations of ammonia to describe the leachate. See Table 3.4 at end of chapter for summary.

The low pH is due to the production of simple organic acids and the production of CO_2 which dissolves in the leachate thus lowering the pH. The high organic loads are also due to the low molecular weight organic acids. Using the $BOD_5:COD$ ratio is an attempt to get some measure of the biodegradability of the leachate. BOD_5 is a test which measures bacterial action on the leachate whereas the COD is a chemical technique which oxidises most of the components in the sample.

The COD is usually larger than the BOD₅, but if the BOD₅: COD ratio is of the order of 0.5 then the leachate is considered to be relatively biodegradable (Crawford, 1985). Many of the simple acids which in the methanogenic phase are converted to methane are present in acetogenic leachate, thus the high ratio. Due to the aggressive chemical nature and low pH of the leachate it has dissolved in it high concentrations of metals (Gettinby et al 1996).

3.5.3 Methanogenic Leachate

Robinson and Gronow (1993) define methanogenic leachate as having chloride concentrations of over 500 mg/l and containing less than 200 mg/l volatile fatty acids. Ehrig and Scheelhaase (1993) defines it as having a high pH and a BOD₅:COD ratio of less than 0.1. This leachate is often referred to as stabilised, but at this stage the landfill is biologically at its most active (Gettinby et al, 1996). The leachate however still contains high ammoniacal nitrogen due to the action of the acetogenic bacteria. This leachate can cause salinity and mineralisation of water it is released into, although from a pollution potential or environmental threat perspective it is seen as the favourable stage for the leachate. See Table 3.4 at end of chapter for summary.

3.5.4 Leachate Composition

Table 3.4 (at end of chapter) shows some indicator parameters and their typical concentrations in both the acetogenic and methanogenic stages. As can be seen above there are no typical values for leachate quality, only broad trends. Applying these trends is however difficult due to the site specific nature of leachate quality. If however one took a leachate quality analysis from a specific site it is possible, by comparing it with the broad trends above together with gas data, to determine at what stage in the decomposition process the refuse is, providing another window into the black box of landfill biochemistry. Any leachate escaping into the environment will impact on the water quality in the

area. Obviously methanogenic leachate will impact less per volume than acetogenic leachate, but it will still impact. The next section will follow what happens when leachate is released into the environment.

3.6 ENVIRONMENTAL IMPACT OF LANDFILL LEACHATES

3.6.1 Introduction

In assessing the impact upon water from landfills we are concerned about potential impacts on both groundwater and surface waters. While considering the impacts upon these resources we must also keep in mind their importance as well as their vulnerability. If surface waters become polluted the result can be the killing of fish, loss of aesthetic value and a general change of the flora and fauna associated with the system. In the case of groundwater, the contamination of the subsurface environment may continue unnoticed for long periods of time with the pollution only noted if the groundwater is abstracted from wells or if it emerges in a surface discharge (Crawford and Smith, 1985). Once the problem is detected however there is a large reservoir of pollution that requires cleaning up. The major aspect of the prevention of pollution by leachate is the prevention of leachate production. Although it is the quality of leachate that causes pollution, it is the quantity of leachate that is most controllable (Crawford and Smit, 1985).

3.6.2 Measurement of Pollution

The characteristics used to measure pollution potential in leachates or waste waters can be broadly divided into four categories: physical characteristics; inorganic chemicals; organic chemicals and toxicity.

The physical characteristics of leachate such as quantity of dissolved solids and suspended solids, temperature, colour, and quantities of inorganic chemicals such as iron and ammonia present are all readily measured by various standard

techniques. However, the measurement of organic pollution is less straight forward and is increasingly based on the oxygen demand of the leachate. Techniques for measuring oxygen demand can be divided into three categories (Crawford and Smith, 1985): biochemical; chemical and instrumental.

Biochemical techniques use bacteria to oxidise the organic matter and the loss of oxygen due to the bacterial activity can be measured as in the biological oxygen demand (BOD). It is expressed as the milligrams of oxygen required by the microorganisms to oxidise a liter of sample over a standard number of days (usually five). Chemical techniques use chemical agents to oxidise the organic material and the difference between the original concentrations of oxidising agents and that which remains after a given contact time with the sample is a measure of the oxygen demand of that sample. The most common chemical technique is the chemical oxygen demand (COD). Instrumental techniques use thermal oxidation of the organic material with subsequent measurement of the gasses produced. A commonly used technique is the total organic carbon (TOC).

Toxicity of a chemical is usually expressed as the concentration of a chemical required to kill 50% of a test organism in a given period. This value is known as the 50% lethal dose, LD_{50} or median tolerance limit, TL_m .

3.6.3 Surface Water Pollution

The capacity of an aquatic system to receive and assimilate contaminants and pollutants is a function of both the aquatic ecosystem and the nature of the chemical, physical and biological properties of the contaminants (Petts and Eduljee, 1994). Some pollutants are biodegradable, but can affect aquatic life indirectly because their breakdown depletes the water of oxygen. Generally the ability of water to absorb contaminants is related to the reaeration capacity of the system i.e. the capacity to absorb atmospheric oxygen. Flowing waters have the ability to a far greater extent than static waters as turbulence enlarges the water-air

interaction area. A reduction in the dissolved oxygen levels will ultimately result in the extinction of oxygen dependent flora and fauna in the system (Petts and Eduljee, 1994; Waste management paper no. 26, 1988; Dallas and Day, 1993; Crawford and Smith, 1985).

If the leachate enters at low pH i.e. acetogenic phase most metals will be more mobile (Baccini, 1989) and hence more bioavailable. These metals will bioaccumulate, although their toxicity varies according to the chemical characteristics of the water (Petts and Eduljee, 1994). Ammonia can be present in landfill leachate up to several hundred mg/l, whereas unpolluted rivers have a very low ammonia content. This ammonia has a threefold effect on a river. Firstly it exerts an oxygen demand on the receiving waters as it is biochemically oxidised to nitrate. Secondly it is toxic and thirdly it is a fertiliser and will alter the ecology of the fauna and flora in the receiving waters. Ferrous iron, Fe^{2+} , is particularly objectionable in a river since ferrous compounds can produce "ochre" deposits, stringy deposits containing bacteria and ferric hydroxide (Crawford and Smith, 1985).

3.6.4 Groundwater Pollution

When leachate leaves the base of a landfill and enters the soil and finally the soil water zone it is subject to processes of attenuation. Attenuation is used as a common term for processes that reduce the concentrations of dissolved contaminants in the leachate plume, including physical (dilution), physiochemical (sorption, ion exchange), chemical (precipitation) and microbial (degradation) (Christensen et al, 1994; Waste paper no. 26, 1988; Crawford and Smith, 1985).

All compounds in leachate entering an aquifer will be subject to dilution as the leachate mixes with the groundwater. For the non-reactive compounds of which chloride is the dominant component, dilution is the only attenuation mechanism. Factors affecting this dilution relate to the nature of the leachate. The viscosity of

leachate may deviate from the viscosity of groundwater (Christensen et al 1994). The density of leachate is a function of temperature and concentration of dissolved solids. Leachate with a total dissolved solids concentration of 20 000 mg/l is not uncommon, this corresponds to a density of over 1 % higher than groundwater. The density difference has a significant effect on the vertical positioning of the plume just below the landfill. Dispersion is a term that accounts for the mixing according to concentration gradients and is represented by the mathematical term dispersivity. Dispersivity has a longitudinal component (in the flow direction) as well as a vertical, horizontal and transverse component. The net result of the above is that most plumes of leachate below landfills is estimated to have been diluted in the order of 100 times at the furthest part of the plume based on chloride concentrations (Christensen et al, 1994).

The entry of strongly reduced landfill leachate, rich in organic matter, into a presumably aerobic aquifer creates a complex environment due to redox processes, biodegradation, dissolution/precipitation, complexation, ion exchange and adsorption processes. These processes create a sequence of redox zones downgradient of the landfill. These redox zones strongly influence the attenuation of contaminants in the plume. The main process responsible for the development of a redox zone sequence in the leachate plume is the microbial degradation of leachate organic matter according to available oxidants (electron acceptors). The easily degradable organic matter will degrade utilising, in sequence, free oxygen and nitrate in the groundwater, followed by oxidised manganese and iron compounds associated with the sediment. Continued degradation may utilise sulphate, and finally methane may be produced by fermentation and carbon dioxide reductions. If leachate is continually released, a methanogenic zone will be developed close to the landfill, where the most easily degradable organic matter is found. According to the thermodynamics of the redox processes, zones of sulphate reduction, ferric reduction, manganic reduction, and nitrate reduction will develop downstream from the landfill if these electron acceptors are present in the aquifer.

The oxidation capacity of an aquifer unit, including solids and water, can be calculated as:

$$\text{OXC} = 4[\text{O}_2] + 5[\text{NO}_3^-] + [\text{Fe}^{3+}] + 2[\text{Mn}^{4+}] + 8[\text{SO}_4^{2-}] + 4[\text{CH}_2\text{O}]$$

(Christensen et al, 1994)

All concentrations are in moles per m^3 . The oxygen, nitrate and sulphate are found in the groundwater, whereas organic matter (represented by CH_2O) and the iron and manganese species are associated with the aquifer solids. A pictorial example from the Vejen Landfill (Christensen et al, 1994) is shown in Figure 3.6.

A further mechanism by which the leachate is treated as it passes through rock or soil is mechanical filtration. The degree of this filtration depends on the sizes of the pore spaces in the soil or rock strata (Crawford and Smith, 1985).

Thus the pollutants in leachate can be attenuated during the passage through soils and rock underlying the landfill by a variety of mechanisms. The extent of attenuation reflects the nature of the soil and rock as well as the quality and quantity of the leachate. A decrease in the passage rate of the leachate through the soil or rock before it encounters ground or surface water reduces the pollution potential. A significant unsaturated zone beneath the landfill provides considerable potential for leachate attenuation before it reaches the groundwater (Christensen et al, 1994).

3.7 CONCLUSIONS

A landfill should be considered as a bioreactor in intimate contact with the environment in which it exists. Waste material is disposed of in the landfill and is subject to biological degradation which in turn produces decomposition products. These waste products leave the landfill either as leachate or as gas and have an

impact on the natural environment in which the landfill exists. The extent of the pollution caused by this is as much a function of the type of waste entering the landfill as the management and control that is exercised in the landfill operations.

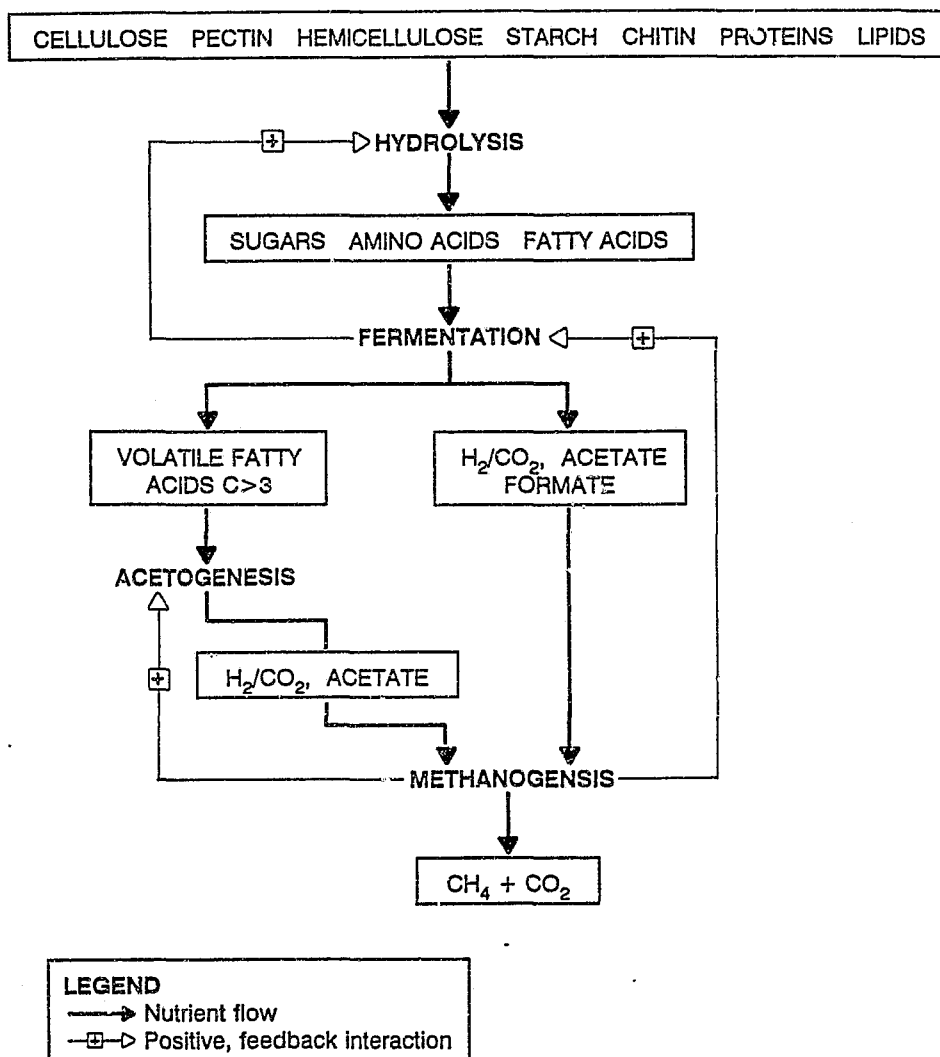
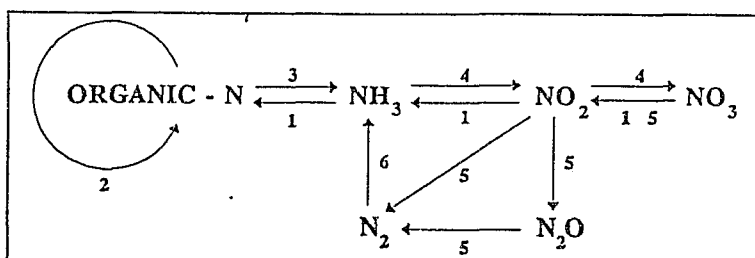


Figure 3.1 Scheme of carbon flow and interactions in the methanogenic syntrophic association (Baccinni, 1989)



1. Nitrogen assimilation (primarily nitrate and ammonia) by plants;
2. Heterotrophic conversions of organic nitrogen from one organism to another;
3. Ammonification, the breakdown of organic nitrogen to ammonia in decomposition;
4. Nitrification, the aerobic oxidation of ammonia to nitrite and nitrate by bacteria;
5. Denitrification, the anaerobic conversion of nitrate to nitrite and ammonia by bacteria and fungi; and
6. Nitrogen fixation, the biological fixation of molecular nitrogen to produce ammonia, which is, in turn, converted to organic nitrogen compounds in organisms.

Figure 3.2 Simplified diagram of the Nitrogen cycle (Campbell, 1996)

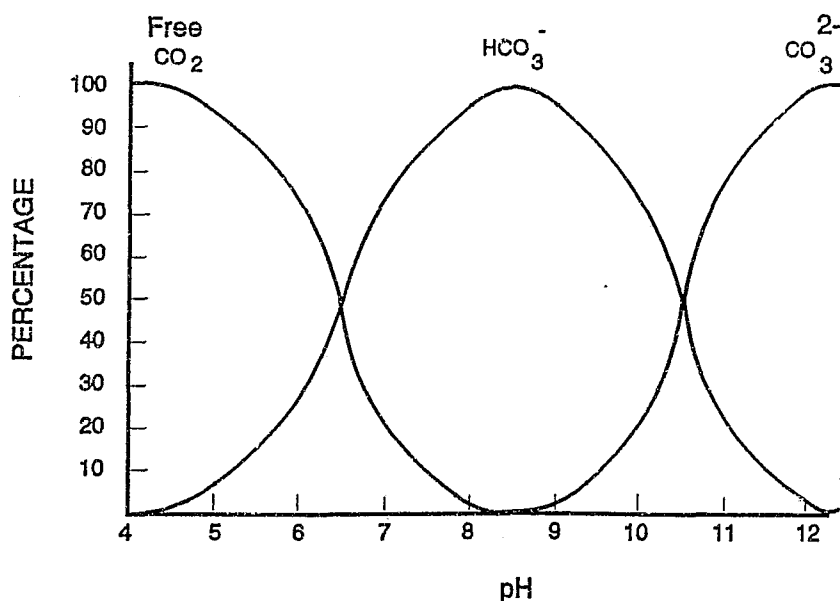


Figure 3.3 The relationship between pH and the percentage of free carbon dioxide, bicarbonate and carbonate ions in water (Dallas and Day, 1993)

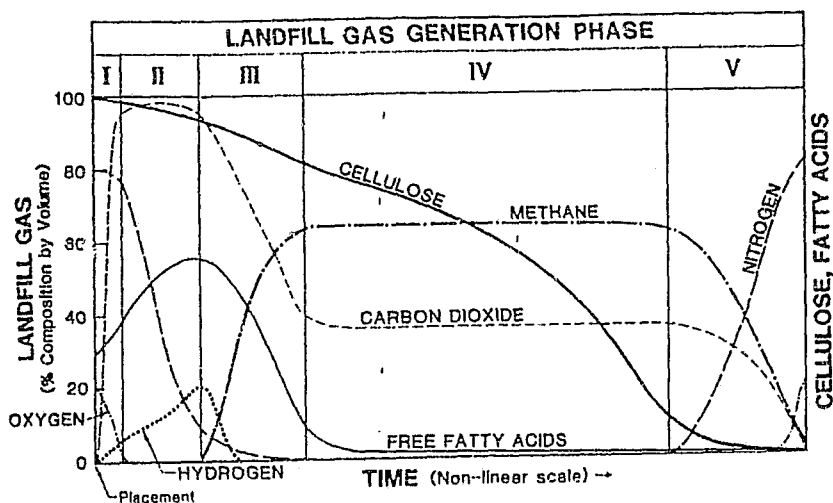


Figure 3.4 Typical landfill gas generation pattern (Augenstein and Pacey, 1991)

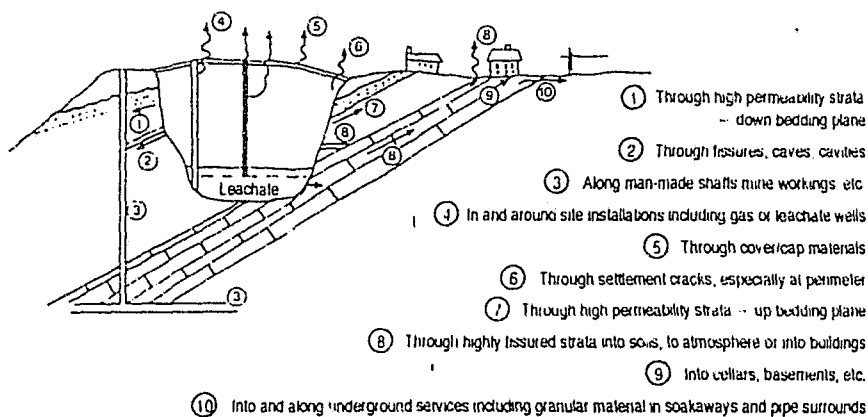


Figure 3.5 Some potential gas migration pathways (Petts and Eduljee, 1994)

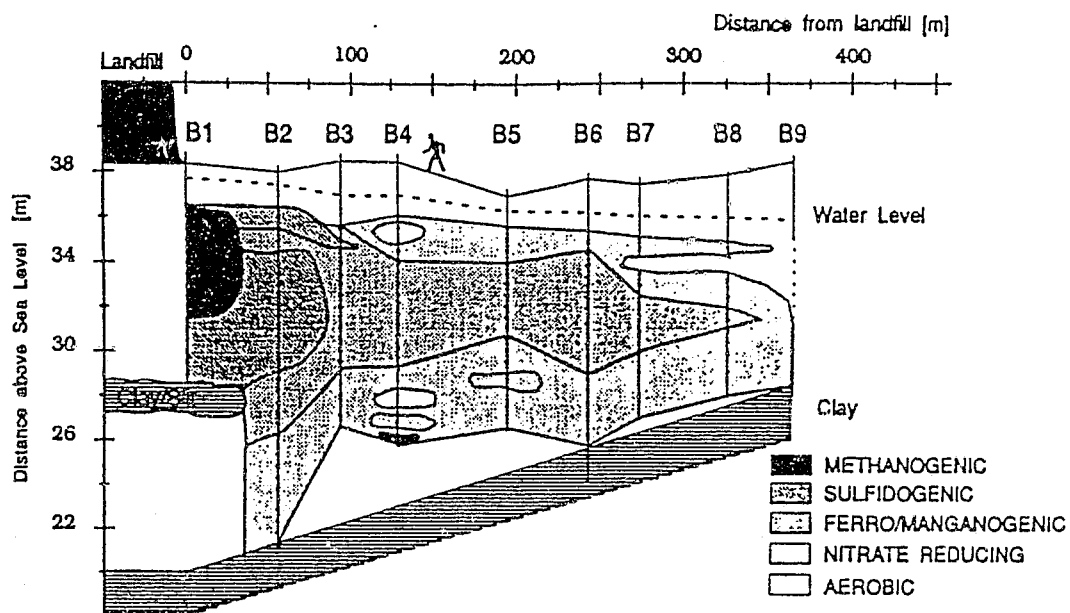


Figure 3.6 Redox zones identified in the Vejen landfill plume (Christensen et al, 1994)

Table 3.4 Summary of leachate compositions (Gettinby et al, 1996, Ehrig and Scheelhaase, 1993, Crawford and Smith, 1985, Petts and Eduljee, 1994)

Parameter	Unit	Gettinby et al and Ehrig			Crawford		Petts		
		A Ave	A Range	M Ave	M Range	< 2 yrs	> 10 yrs	Acetogenic	Methanogenic
pH		6.1	4.5-7.5	8.0	7.5-9.0	5.0-6.5	6.5-7.5	5.0-6.5	6.5-8.0
BOD ₅	mg/l	13000	4000-40000	180	20-550	4000-30000	<100	20-40000	50-1000
COD	mg/l	22000	6000-60000	3000	500-4500	10000-60000	50-500	25-60000	1000-5000
BOD ₅ /COD		0.58		0.06				>0.5	<0.5
SO ₄ ²⁻	mg/l	500	70-1750	80	10-420	50-1000	<10		
Ca	mg/l	1200	10-2500	60	20-600	500-2500	100-400		
Mg	mg/l	470	50-1150	180	40-350				
Fe	mg/l	780	20-2100	15	3-280	100-1500	10-400	several hundred	<50
Mn	mg/l	25	0.3-65	0.7	0.03-45				
Zn	mg/l	5	0.1-120	0.6	0.03-4			10-100	<1
Cl	mg/l	2100	100-5060	These parameters					
Na	mg/l	1350	50-4000	not affected by		500-2000	100-500	1500-3000	1500-3000
K	mg/l	1100	10-2500	phase		500-3000	<200		
Alkalinity	mg/l as CaCO ₃	6700	300-11500			200-1000	50-400		
NH ₄ ⁺	mg/l	750	30-3000					5-15000	3-10000
Total N	mg/l	1250	50-5000					500-1500	500-1500
NO ₃ ⁻	mg/l	3	0.1-50			100-1000	<100		
NO ₂ ²⁻	mg/l	0.5	0-25						
Total P	mg/l	6	0.1-30						
TOC	mg/l					1000-20000	<100	10-25000	300-1500

Along with the Minimum Requirements for Waste Disposal by Landfill the investigation will focus on whether waste composition with respect to socioeconomic levels can be used to further increase the affordability of waste disposal.

4.1 SUPPORTING THEORIES

The critical variable affecting the composition of waste generated by a community is the socioeconomic condition of the community (Gibbons et al, 1992). Wealthy communities form "throw away" societies, whereas poorer communities have less to throw away and are more ingenious in reusing and refurbishing articles that a wealthier community would discard (Blight et al, 1994). These factors, together with the methods of collection used and the use of coal and wood burning stoves for heating and cooking in certain areas, results in very different waste compositions between developing and developed communities, see Table 4.1. Generally waste from developed areas has a high paper and organic putrescible component. Waste from developing communities that are not supplied with electric power tend to have a lower paper and organic putrescible component but a very high ash and dust component. The dust is mostly a result of yard sweeping and of dumping waste on street corners, so that when it is collected front end loaders are needed. These then tend to pick up significant amounts of soil and dust with the waste.

Table 4.1 Composition of municipal refuse in developed and developing countries (Blight, 1994)

Composition in % by mass	Developed Countries		Developing Countries		Mixed Economies	
	USA	UK	India (Delhi)	China (Wuhan)	South Africa	
					Soweto	Johannesburg
Vegetable	22	25	47	16	9	20
Paper	34	29	6	2	9	36
Metals	13	8	1	0.5	3	10
Glass	9	10	0.6	0.6	12	11
Textiles	4	3	-	0.6	1	1
Plastics	10	7	0.9	0.5	3	7
Wood	4	18	44.5	1.8	63	15
Dust, ash and others	4			78		
Refuse density kg/m ³ (uncompacted)	100+	150	420	600 (estimated)	400	170
Refuse generation rate Tons/person/year	0.65	0.65	0.14	0.20	0.15	0.58 to 0.73

Most work on refuse decomposition and subsequent leachate and gas quality and quantity has been executed in developed countries, using waste typical of that country. The waste studied therefore tends to have very high putrescible and paper components, usually well over 50% by wet weight (Farquhar and Rovers, 1973, Rees, 1980, Lee et al, 1993, Owens et al, 1992). It is felt that the lower putrescible component in developing communities waste will produce less concentrated leachate (Blight, 1994; Kabbe and Rohrs, 1994) and therefore have less of a pollution potential. Another positive aspect is the significantly lower per capita waste generation rates in developing communities. Of greater importance than the lower organic component is the high ash and dust component. The ash results from the burning of solid fuels for cooking and heating in areas that are not electrified,

although from personal experience even areas that are electrified still have a significant ash component in their waste stream. Verrier (1983) stated that electrification of a township did not result in a significant reduction of the ash content. The fuels burned include the traditional coal (all grades) and wood, as well as components of the waste stream like paper and even plastic. Work on the co-disposal of power station coal ash and refuse (Cossu et al, 1991; Kabbe and Rohrs, 1994) showed that the ash accelerates the biodegradation of the organic fraction of the waste and that the methanogenic phase is established sooner, along with its obvious advantages.

The ash component achieves this by increasing the pH and alkalinity as CaCO_3 , thus providing a favourable environment for the establishment of methanogenic bacteria. Rees (1980) also showed that a high density was needed to establish methanogenesis in municipal solid waste. It is important to note that even loose waste from developing communities tends to have high densities. It is unknown at this stage what effect the ash will have on the Eh (Redox Potential) and the field capacity or moisture retention capacity of the waste, both of which are important in terms of refuse decomposition. Another aspect which is as yet unknown is the ability of the ash to immobilise toxic compounds, specifically heavy metals. Gavasci et al (1992) studied this by allowing leachate to run through natural pozzolanic pyroclastic soils and achieved favourable results.

It seems that waste from developing communities has a few significant differences from waste that occurs in a developed community. The effect of these differences on biodegradation has remained largely uninvestigated but it seems as if they may have a significant influence on the polluting potential of waste. If waste composition does affect polluting potential then the standards required for landfilling of different wastes can be adjusted to make waste disposal more affordable to developing communities.

4.2 FORMAT OF INVESTIGATION

The comparative study of the decomposition processes and products of developed and developing communities waste was divided into three main sub-problems with specific questions that needed to be addressed in each.

- How does waste composition vary across socioeconomic levels?

This section required the identification of waste sources typical of their respective communities and the classification of these wastes, see Chapter 5.

- How does waste composition affect polluting potential?

This section comprises the majority of work undertaken and entailed finding the answers to a number of important questions :

How does composition affect refuse degradation and its variation with time i.e. total pollution potential?

How does refuse composition affect leachate quality and quantity, as well as landfill gas generation?

Is the ash component significant in decomposition as well as in heavy metal immobilisation?

In order to answer these questions, selected refuse was allowed to degrade and the products of that decomposition analysed in order to obtain an understanding of the processes involved. This study made use of field cells, test cells and lysimeters, see Chapters 6, 7, 8 and 9.

- Does a different pollution potential require different landfilling standards?

This section deals with how landfilling practice may be modified with the results from the investigation to make it more economically affordable to the community it serves, see Chapter 10.

5.1 AIM

The first step taken when starting the investigation was to determine if there was any significant difference in waste composition between developed and developing communities. Previous work (Rushbrook et al, 1988), (Gibbons et al, 1992), (Malan et al, 1988), (Stewart Scott, 1991), seem to indicate a trend, although these results were all from different areas and were taken at different times. One of the critical factors identified from the start was to find suitable sources of solid waste that were representative of their origins. The waste source had to fulfil a number of requirements, namely:

- Developing community waste had to be from a large informal area, preferably not electrified, hereafter referred to as poor waste.
- Developed community waste had to be from a formal middle to upper income residential area, hereafter referred to as rich waste.
- Both sources had to be in close proximity (geographical) and arrive at a central location where the wastes could be studied.
- There had to be easy access to sampling of the waste.
- Emphasis had to be placed on safety aspects.

After extensive research and with the co-operation of the Eastern Services Council (ESC), a source was identified at the Heidelberg Transfer Station. This transfer station accepts waste from the town of Heidelberg as well as the township of Ratanda. Due to previous apartheid policies, the two areas are geographically very close together yet there is a vast difference in their socioeconomic levels. In a previous study conducted by Scott and DeWaal, the following figures are quoted (Scott and DeWaal, 1991).

Heidelberg supports a population of 15000 people with an average population density of 13 people per hectare. The average population in Heidelberg falls into the medium income group with a small low income contingent. Refuse is generated at a rate of 1.6 kg/person/day at a density of 200 kg/m³. Ratanda supports a population of 35000 with an average population density of 118 people per hectare. The average population of Ratanda falls into the sub-economic income group. Refuse is generated at a rate of 0.4 kg/person/day at a density of 340 kg/m³. The township is characterised by two large informal settlements and relies mainly on solid fuel burning for cooking and heating.

5.2 STUDY METHODS

In order to clarify the data above and to get a hands-on feel of what difference there is in the waste composition, it was decided to do a source classification on both towns at the point where the waste arrives at the transfer station. From the available literature (Leroy et al, 1992), (Brunner, 1986), (CRC handbook, 1973), a list of 9 categories was compiled in which to differentiate the waste, namely: ferrous metals; non-ferrous metals; glass; textiles; plastics; paper; vegetable and putrescible; dust and ash and miscellaneous (including builders rubble, ceramics, unburnt coal, leather, rubber etc). After an initial period of trial and error a sampling method was developed that was not time consuming and did not interfere with the running of the transfer station.

The waste from Ratanda, which arrives loose at the station, was collected into two to three 35 l buckets. This gave a mass of approximately 20 to 30 kg for each sampling run. The waste was sieved through a 15.8 mm sieve into the dust and ash container, the rest being hand sorted into the other 8 categories. The choice of the 15.8 mm sieve was mainly due to time, although nearly 80% of ash passing the 15.8 mm sieve passed through the 4.75 mm sieve, the balance being mainly unburnt coal, feathers and cigarette ends. The buckets with the different categories were then weighed and the

composition determined on a wet weight basis. The Heidelberg waste arrived in refuse bags in a 19 m³ compactor. It was decided to classify three to four 85 l bags of refuse per load. Only unbroken bags were chosen for analysis. The bags were taken to the sorting area, broken open and their contents sorted.

One of the main objectives of the classification was to determine how the ash content varied over a full year. A seasonal variation was suspected, so classifications were conducted every three months for a year. For each of these, one week was spent on site so that the classification for a season was an average of five days in that season. Historical data from the weighbridge was also studied so that generation rates could be confirmed. See Table 5.1 and Figures 5.1 and 5.2.

A further classification was conducted in October 1995 to act as a check for the previous work as well as providing a real time comparison for a classification of waste from Watville and Benoni, the waste which was used in the laboratory and field studies. The Watville and Benoni wastes were classified at the Eastern Services Council's Weltevreden landfill site.

Table 5.1 Summary of refuse compositions (percentage wet weight) and comparisons with other sources

FERR	NON-FE	GLASS	TEXT	PLAST	PAPER	PUTRES	ASH	MISC	ORIGIN	REF
POOR WASTE										
2.0		2.0		3.0	3.0	2.0	84.0	4.0	EAST RAND	STEWART SCOTT 1991
1.6		2.0		2.4	3.4	4.6	82.0	2.4	EAST RAND	GIBBONS 1992
1.4	0.1	1.3	0.7	2.1	1.4	9.4	78.8	5.0	RATANDA	JUNE 1994 OWN
1.0	0.2	2.0	0.7	3.2	1.9	16.2	69.5	5.1	RATANDA	OCTOBER 1994 OWN
2.1	0.4	1.3	1.3	4.8	5.2	28.8	54.0	2.1	RATANDA	FEBRUARY 1995 OWN
1.3	0.3	1.6	0.5	1.8	2.3	17.4	71.5	3.3	RATANDA	MAY 1995 OWN
0.8	0.7	1.7	0.4	4.0	4.4	26.2	56.5	5.3	RATANDA	OCTOBER 1995 OWN
2.2	1.2	2.1	1.8	6.1	4.0	17.6	56.8	8.2	WATTVILLE	OCTOBER 1995 OWN
1.0		1.0		1.0	6.0	47.0	44.0		INDIA	RUSHBROOK 1988
0.5		0.6	0.6	0.5	2.0	16.0	78.0	1.8	CHINA	RUSHBROOK 1988
2.5		0.6	1.4	4.0	6.6	76.0	8.9		NIGERIA	CALREC. 1994
0.1		0.2	3.0	1.0	2.0	74.7	19.0		INDIA	CALREC. 1994
RICH WASTE										
7.0		10.0		11.0	20.0	29.0	9.0	14.0	EAST RAND	STEWART SCOTT 1991
6.0		4.0		17.0	37.0	32.0	0.6	6.0	EAST RAND	GIBBONS 1992
8.7	0.9	14.7	1.1	11.6	21.1	41.1	0.2	0.7	HEIDELBERG	JUNE 1994 OWN
7.7	2.2	7.2	1.6	13.3	23.7	41.4	0.0	2.9	HEIDELBERG	OCTOBER 1994 OWN
9.1	3.4	2.8	0.7	12.7	28.1	42.8	0.0	0.4	HEIDELBERG	FEBRUARY 1995 OWN
5.7	7.3	5.7	0.4	12.6	21.1	45.4	0.0	1.8	HEIDELBERG	MAY 1995 OWN
1.3	4.8	9.4	1.2	15.5	22.8	44.2	0.0	0.6	HEIDELBERG	OCTOBER 1995 OWN
5.3	3.2	12.0	0.5	7.2	24.2	47.0	0.0	0.6	BENONI	OCTOBER 1995 OWN
6.8	3.2	10.3	1.9	14.4	21.8	40.1	0.0	1.5	SANDTON	AUGUST 1994 OWN
13.0		9.0	4.0	10.0	34.0	22.6	4.0	4.0	USA	RUSHBROOK 1988
8.0		10.0	3.0	7.0	29.0	25.0	18.0		UK	RUSHBROOK 1988

5.3 RESULTS

The Ratanda data tended to agree with previous work done in the area (Stewart Scott, 1991), (Gibbons, 1992) and with work done in China (Rushbrook, 1988), see Table 5.1 and Figures 5.1 and 5.2. The waste classifications done in India and Nigeria (Rushbrook, 1988, CalRecovery, 1994 pers. comm.) however, showed a different composition with the major components in the refuse being organic putrescible. This difference could be ascribed to climatic differences, fuel supply, cultural differences and possibly even religious differences. A clear seasonal variation in the ash content can be seen, from a peak of 79% of the total in the winter to a low of 54% in mid-summer. The other components of the waste stream tend to stay stable in terms of quantity, although when expressed as a percentage they increase as the ash content decreases. The Heidelberg data also agreed with previous work done in the East Rand area as well as with work done in the United Kingdom and in America. A sample of Sandton waste was classified in August 1994 when prototype lysimeters were commissioned. This data also followed the trend set by other developed areas. Over more than a full year of classification the composition in Heidelberg did not change significantly. The important component in terms of being biodegradable, paper and putrescible, stayed almost constant at around 65% by wet weight. Two significant trends that did emerge however was the increase in non ferrous metals as bimetallic beer and soft drink cans were replaced by aluminium ones. Also significant was the decrease in glass which was due to a recycling operation opening in town (pers. comm.).

Generation rates computed from the historical data were 0.36 kg/person/day for Ratanda and 1.2 kg/person/day for Heidelberg. Loose densities were approximately 400 kg/m³ for Ratanda waste and 120 kg/m³ for Heidelberg.

As it is felt the ash component has a large influence on refuse decomposition, a range of tests were performed on this component once it had passed a 4,25 mm sieve. Firstly a

chemical analysis was performed on the June 1994 and October 1994 samples, see Table 5.2, the other sampling runs were not analysed due to the costs involved. As expected the ash had a high pH which should help to establish rapid methanogenesis. The ash also contains significant amounts of Phosphorous and Nitrogen which are needed as macronutrients by the methanogenic bacteria. Among other trace metals there were very high levels of Iron, Aluminium and Calcium. The effect that these will have is as yet unknown however, although it is suspected they could influence alkalinity.

Table 5.2 Chemical analysis of ash passing 4.75 mm sieve.

Parameter	Unit	June 94	October 94
Volatile matter	g/kg	32	69
PH	-	8.1	7.9
Ash	g/kg	766	930
Total Phosphorus as P	mg/kg	4 310	2 550
Kjeldahl Nitrogen as N	mg/kg	6 350	2 000
Iron as Fe	mg/kg	23 000	25 000
Copper as Cu	mg/kg	29	29
Zinc as Zn	mg/kg	400	300
Cadmium as Cd	mg/kg	<3	<3
Mercury as Hg	mg/kg	0.34	<8
Lead as Pb	mg/kg	40	600
Aluminium as Al	mg/kg	28 600	24 100
Calcium as Ca	mg/kg	51 100	19 000
Magnesium as Mg	mg/kg	6 000	3 700
Potassium as K	mg/kg	530	1 570
Chromium as Cr	mg/kg	10	140
Specific Energy	MJ/kg	0.42	0.46

To determine particle sizes of the ash, grading and hydrometer tests were performed on the June '94, see Figure 5.3, and October '94, see Figure 5.4, samples. The ash consisted of over 40% fines (< 0.06 mm)(Craig, 1987) in the June sample changing to 30% in the October sample. This fines component could have an important influence on the water retaining ability of the waste and hence influence the field capacity.

5.4 CONCLUSIONS

The classification study provided a good hands-on introduction to the different refuse compositions encountered in developed and developing areas. The results obtained showed a favourable correlation with earlier work as well as tying in with work done at another site. At this stage a definite difference in composition had been established between the two waste types. It was now necessary to establish how this difference affected the processes and products of refuse degradation.

HEIDELBURG REFUSE COMPOSITION PERCENTAGE BY MASS

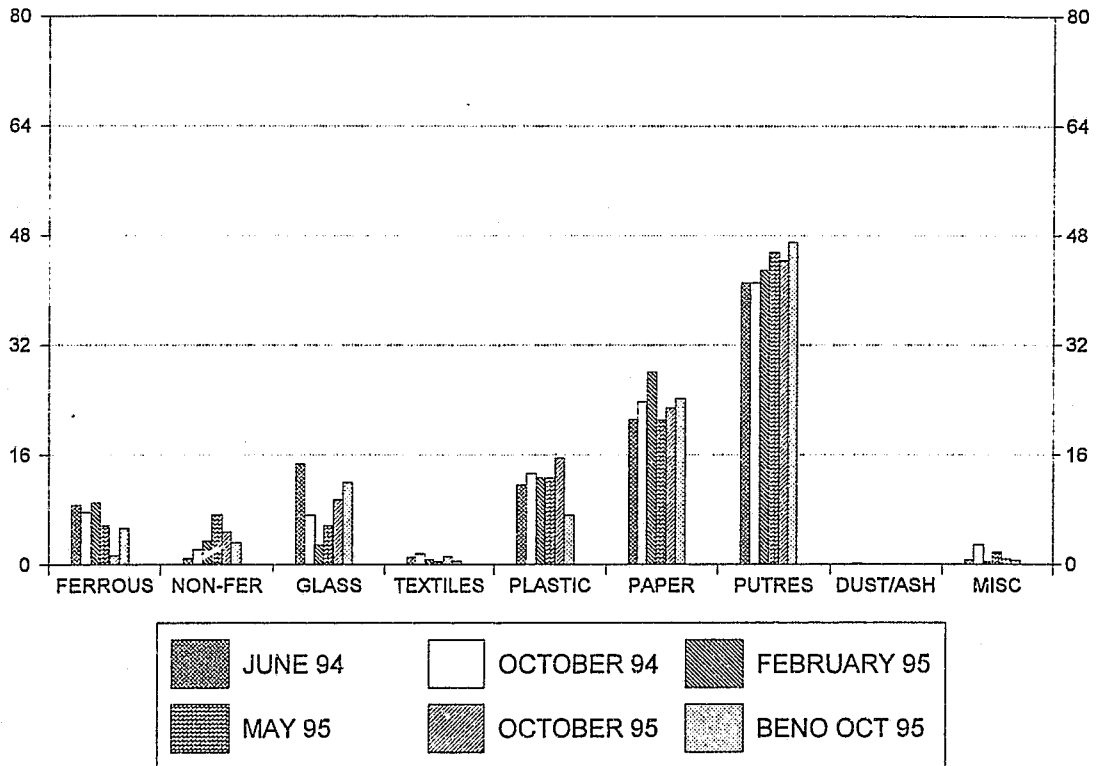


Figure 5.2 Yearly variation of waste from Heidelberg

RATANDA REFUSE COMPOSITION PERCENTAGE BY MASS

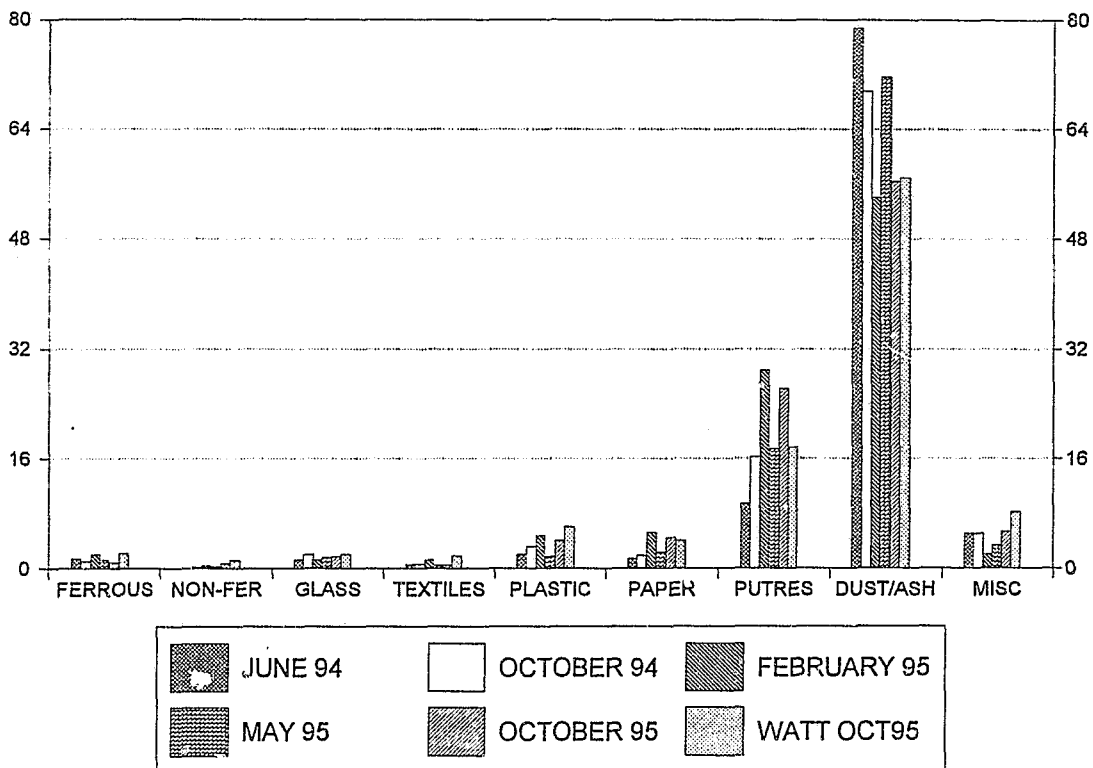


Figure 5.1 Yearly variation of waste from Ratanda

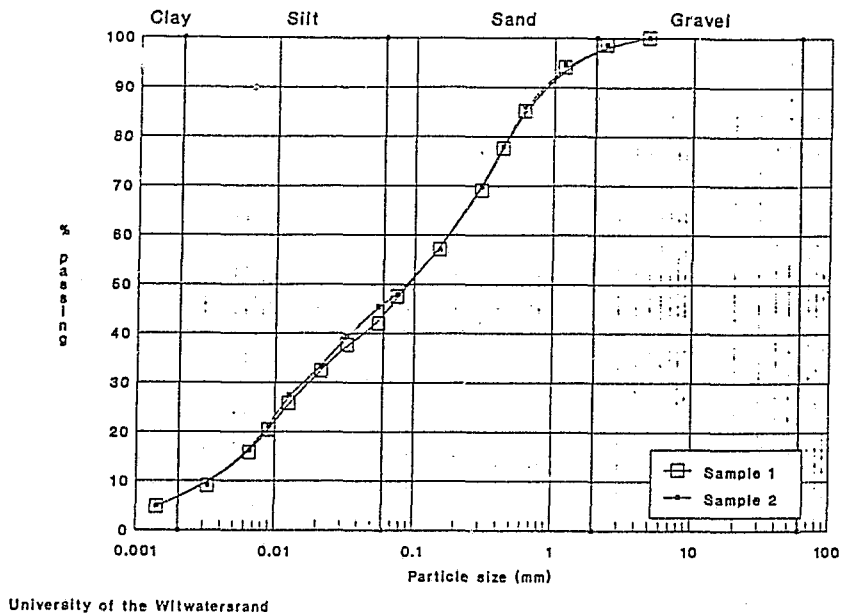


Figure 5.3 Grading analysis of ash component, June 1994

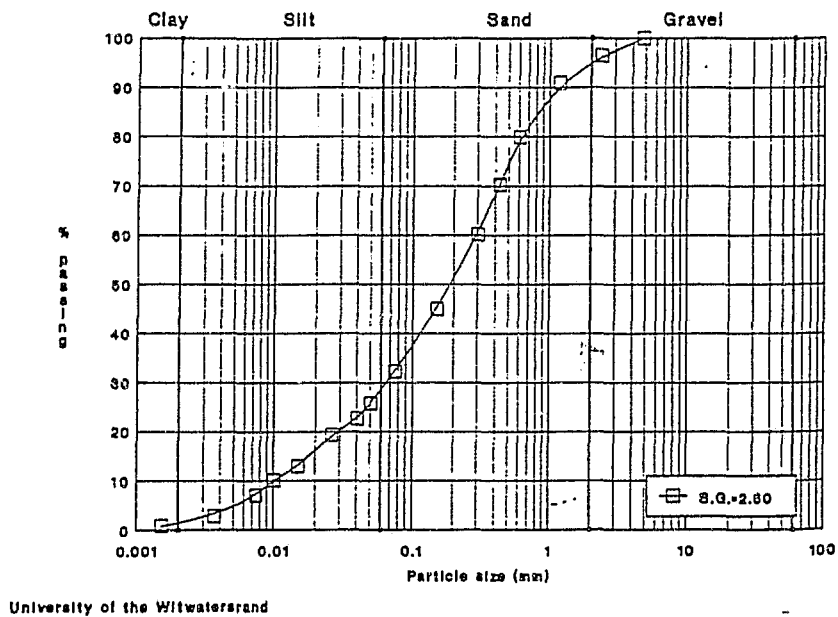


Figure 5.4 Grading analysis of ash component, October 1994



Plate 5.1 Hiedelburg refuse sample

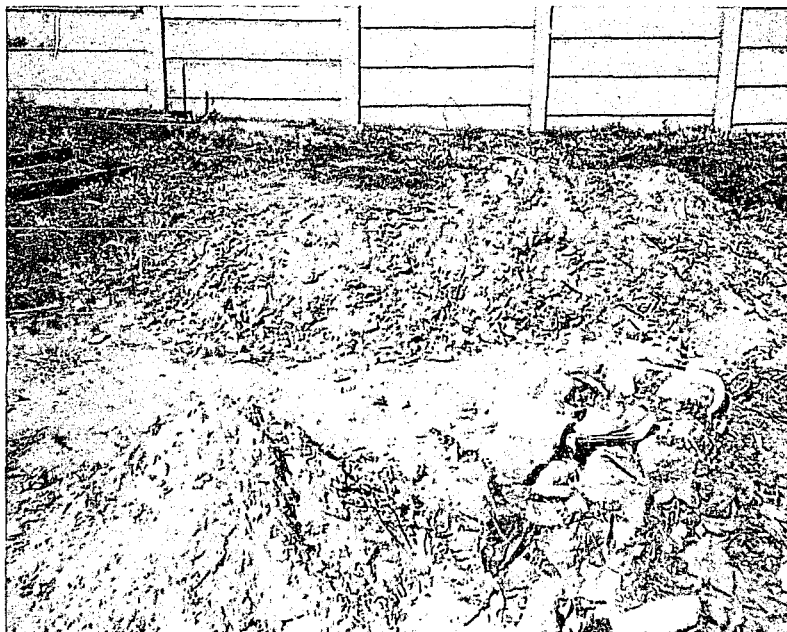


Plate 5.2 Ratanda refuse sample



Plate 5.3 Heidelberg sample showing fraction by volume



Plate 5.4 Ratanda sample showing fractions by volume

Lysimeter studies in the laboratory to model landfill degradation have become accepted practice over years of investigation into landfill behaviour (Cossu et al, 1987, Rees, 1980, Farquhar and Rovers, 1973, Owens, 1992, Kabbe and Rohrs, 1994, Hertig et al, 1993). The system has advantages in that the waste is allowed to degrade in a controlled environment with the products of this decomposition always readily available for analysis. Lysimeter studies were used to answer the main questions in the investigation ie. how does refuse composition affect polluting potential? A typical lysimeter consists of five basic components, namely: a suitable container; an irrigation system; a leachate collection and sampling system; a gas collection and sampling system and a temperature control system.

The container must be airtight, to allow anaerobic conditions to develop, and be made out of an inert material so that the waste and its products do not react with the container. An irrigation system is needed to introduce water into the lysimeter which can then percolate through the waste to form leachate. Obviously no air can be allowed to enter the system during irrigation and the rate of application of water must model the rainfall in the field. Once the leachate has percolated through the waste it should collect in the drainage layer from where it can be sampled. Again no air can be allowed to enter the system. In the headspace left in the drum the gaseous products of decomposition will collect. A gas collection and sampling system is needed so that the gaseous products can be measured. To simulate conditions in a real landfill the temperature needs to be kept at a constant value suitable for bacterial growth. The temperature control system does this by monitoring the temperature and heating the drum when necessary. Even a slight change in temperature can have a large effect on bacterial activity (Farquhar and Rovers, 1973).

6.1 AIM

Previous lysimeter work done in the Department of Civil Engineering used 220 l HDPE drums with a sealable lid. However, on site and in the lab these turned out to be difficult and cumbersome to work with as well as being expensive. The prototypes therefore consisted of one 220 l sealable lid HDPE drum and one 50 l sealable lid HDPE drum. All the other systems on the drums remained the same. The main aim of the study was to see if there was any marked scale effect between the two sizes of drums. If there was no difference then the 50 l drum would be used in the main lysimeter study. The study was also used to see if, with a similar waste composition, similar results to overseas lysimeter studies could be obtained.

6.2 STUDY METHODS

In order to test the design of lysimeters and the associated backup equipment two prototype lysimeters were constructed and filled on the 11th August 1994 with refuse from Sandton (See Table 5.1). The refuse was hand sorted and classified in order to find its composition. To negate scale effects, and to stop preferential pathways developing in the waste, the waste was shredded by hand before it was placed in the lysimeter. The shredded waste was then mixed together and lightly compacted into the drums. The lysimeter design is shown in Figure 6.1.

The irrigation system consisted of a spiralled hose-pipe that was glued to the inside of the drum's lid. Both the ends of the pipe were sealed and 1 mm holes were drilled at 50 mm intervals on the underside of the hose-pipe. While the pipe was being glued onto the lid a hole was drilled into the top of the pipe which was connected to a bulkhead fitting, this in turn was connected to a ball valve on the outside of the lid. A modified syringe connected to an inverted 2 l plastic bottle acted as the water delivery system.

The syringe was screwed into the closed ball valve and the combination held in place by a modified retort stand resting on the lid. The bottle was filled with the required amount of water and then the valve opened. Before all the water had drained out the valve was again closed so that no air was entrained. The system delivered water evenly over the surface at an average flowrate of 20 l/hr. For the purpose of the study distilled water was added to the drums. The water was added at random intervals in the same volumetric ratio to the two drums ie. 0.5 l to the 50 l drum and 2.2 l to the 220 l drum. Leachate was allowed to pond in the leachate collection layer and samples taken at convenient intervals and analysed.

Leachate collects at the bottom of the drum in the leachate drainage layer, which consisted of a 50 mm deep layer of 22 mm washed granite stone, and was then drained off by a bulkhead and ball valve placed at the bottom of the drum. More water was added than leachate removed so that the leachate dammed up in the bottom of the drum, its level being shown by the level indicator pipe. This pipe also assisted in the drainage of the leachate by breaking any airlocks and equalising the pressure in the drum. In order to get the leachate out of a sealed environment a pressure differential was created by the use of a nitrogen filled bag attached to the collection bottle. The sampled leachate was analyzed at the Johannesburg City Council's Cydna Laboratories.

The gaseous products of decomposition collected in the headspace of the lysimeter. Normally these vented off to the atmosphere through a water filled column which again prevented oxygen from entering the system. When the gas was to be sampled the drums were closed off for approximately one hour. The gas was then extracted and captured in a gas bomb. The gas was taken to the Dental Research School at Wits University who were gracious enough to let us use their gas chromatograph. The gas extracted was measured for relative percentages of O₂, N₂, CO₂ and CH₄. The chromatograph used was a Shimadzu GC 8AIT with associated integrator. The unit used a thermal conductivity detector (TCD) running at 130°C with the column and oven set at 30°C.

Two columns were used, a Poropak Q 100/200 mesh to detect CH_4 , CO_2 and a combined peak of N_2/O_2 . A molecular sieve 13X 80/100 mesh was then used to separate out the O_2 and N_2 peaks. The carrier gas used was helium at a flowrate of 25ml/min and the samples were injected with a 0.5 ml precision sampling syringe with "Pressure-Lok" and a side port needle. The results were normalised to 100%.

The heating system consisted of three separate components, two external and one inside the drum. The external equipment consisted of an electric heater tape which was wrapped around the outside of the drum in a spiral. The drum was then wrapped in 50 mm thick glass wool which acted as an insulation. The heater tape was connected to a temperature controller which was set at a specific temperature eg. 30°C . The controller in turn was connected to an external temperature sensor on the outside of the drum. Feedback between these systems kept the drum at a constant temperature. Independent of this system was a thermocouple wire placed directly into the waste and emerging on the outside of the drum through a sealed septum in the lid. The thermocouple was connected to a thermocouple bridge so that a separate reading could be taken of the temperature in the waste. These readings were taken daily at the same time. Due to the way an existing lysimeter study was set up and a lack of spare heater tapes the two prototypes were run at slightly different temperatures. The 220 l drum was run at an average temperature of 23°C while the 50 l was run at an average temperature of 28°C .

6.3 RESULTS

Gas data from the analysis of the two lysimeters over a period of 264 days of monitoring are presented in Figure 6.2 and 6.3. Due to an incorrect sampling method the data for the first 70 days is not valid. During this period the gas was sampled by evacuating the gas bomb under a vacuum of 5 mm Hg and then opening it when it was connected to the gas sampling port. As a result, when a sample was drawn into the gas syringe it was

still under negative pressure and drew in more air. This, combined with the fact that the vacuum was not causing a sufficient flow of gas through the bomb for a representative sample to be captured, led to the method being changed. The new method used a pipette pump to draw gas through the gas bomb at atmospheric pressure. Roughly three times the volume of the gas bomb was drawn through before the gas bomb was sealed. When the original sampling method was used and the gas analysed, it was thought the large drums would have to be used because the gas data was so different. However with long term monitoring it was seen that the gas production patterns were the same, with the time lag being caused by the difference in temperature. What had happened in the prototypes is that the CO₂ bloom occurred, ie. phase 2, and then as the environment in the drum was not conducive to methanogenesis (See Chapter 3) the gas composition steadied off to the values shown with high N₂ and CO₂ and little or no O₂ and CH₄. The low O₂ confirmed that the drums were anaerobic.

Five samples of leachate were taken at periodic intervals and a complete analysis performed on them. It was decided that more regular tests would not be done due to the prohibitive cost involved. Important trends isolated from the data are shown in Figures 6.4, 6.5 and 6.6. As can be seen from Figure 6.4 the pH remained acidic for the duration of the test, COD also remained constant at relatively high values. This indicated that the aerobic and subsequent acid fermentation phase had occurred, however the subsequent methanogenic phases had not been established as with the gas data. The reason for this is most likely the placed density of the waste. Rees (1980) found that under conditions of moisture saturation a density of 500 to 1000 kg/m³ is required to establish methanogenesis. The waste in the prototypes was placed at a density of 210 kg/m³. While a high moisture content, temperature and larger surface area (shredding) promotes microbiological activity a high density allows for the controlled hydrolysis and fermentation of refuse polymers to be effectively coupled to CH₄ production. If the placed density is too loose hydrolysis and fermentation occur too fast, a very high CO₂ bloom occurs (over 90%), and the leachate is acidified to such an

extent that the environment in the lysimeter becomes toxic to the methane producing bacteria. Another significant trend in the two graphs is the time lag of events between the two drums. Again it is felt that this was due to the temperature difference between the two drums. When the data was compared with acidogenic leachate data from other studies a good correlation was found.

6.4 CONCLUSIONS

The prototype lysimeters performed well and a good correlation, although lagged, was found between them. The 50l drums were chosen for use in the full scale laboratory lysimeter study as they were easier to manipulate on site and in the laboratory. Available space in the laboratory also made the use of these drums easier. The prototype study also allowed enough time for familiarisation with the systems and procedures required to monitor the waste degradation in lysimeters. Initial "learning-curve" problems were also addressed so that these did not impact on the full-scale study. The next stage was to answer the main question of the investigation, "How does refuse composition affect the processes and products of refuse decomposition?"

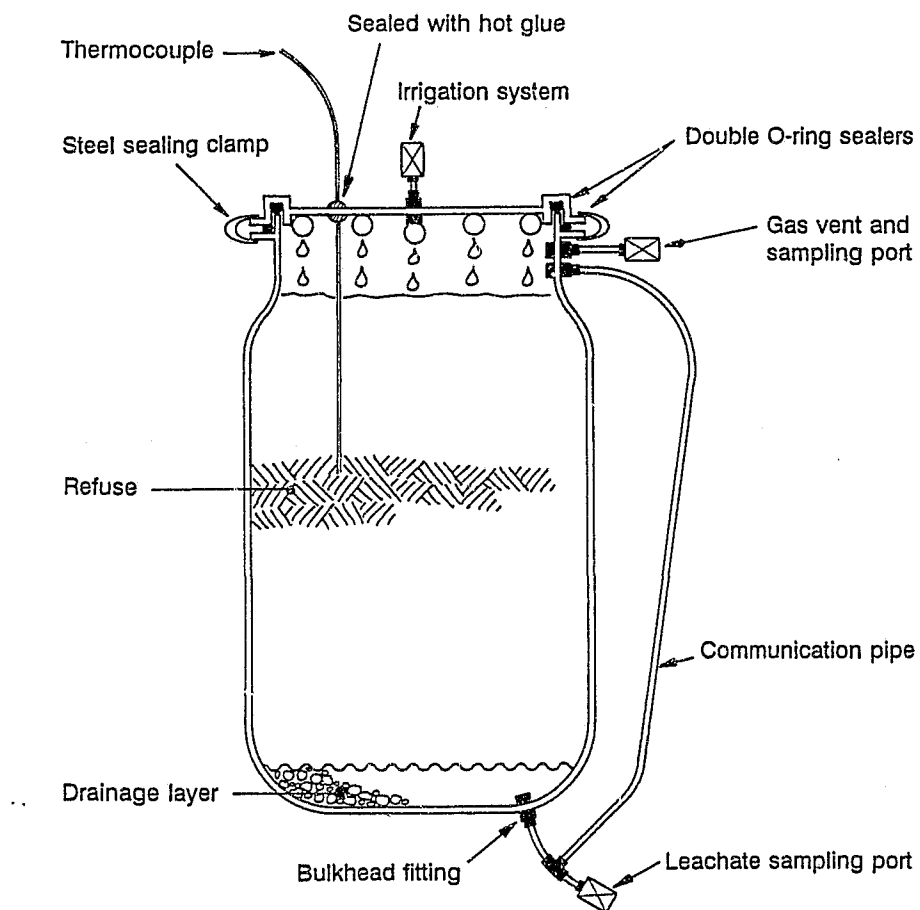


Figure 6.1 Prototype lysimeter design

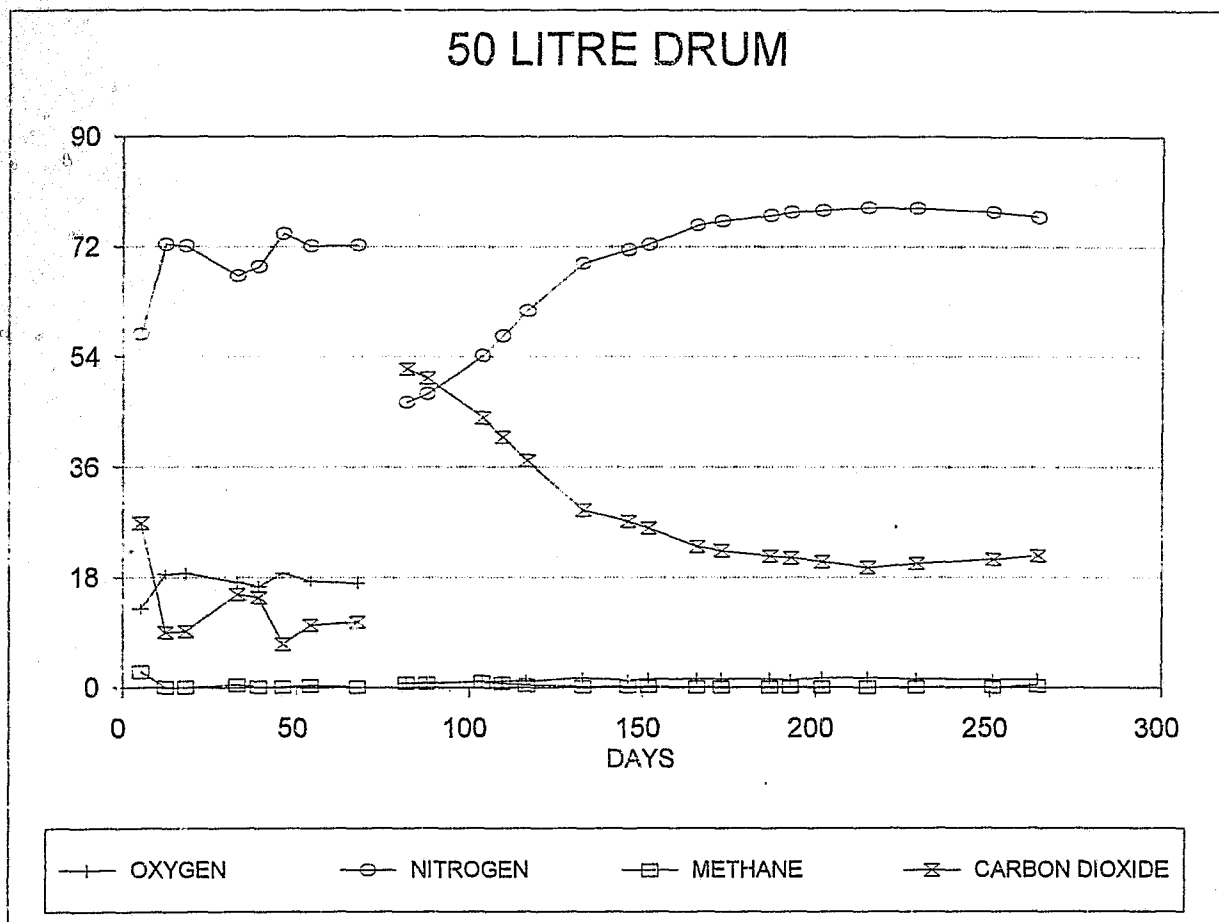


Figure 6.2 Gas analysis for 50 l drum

220 LITRE DRUM

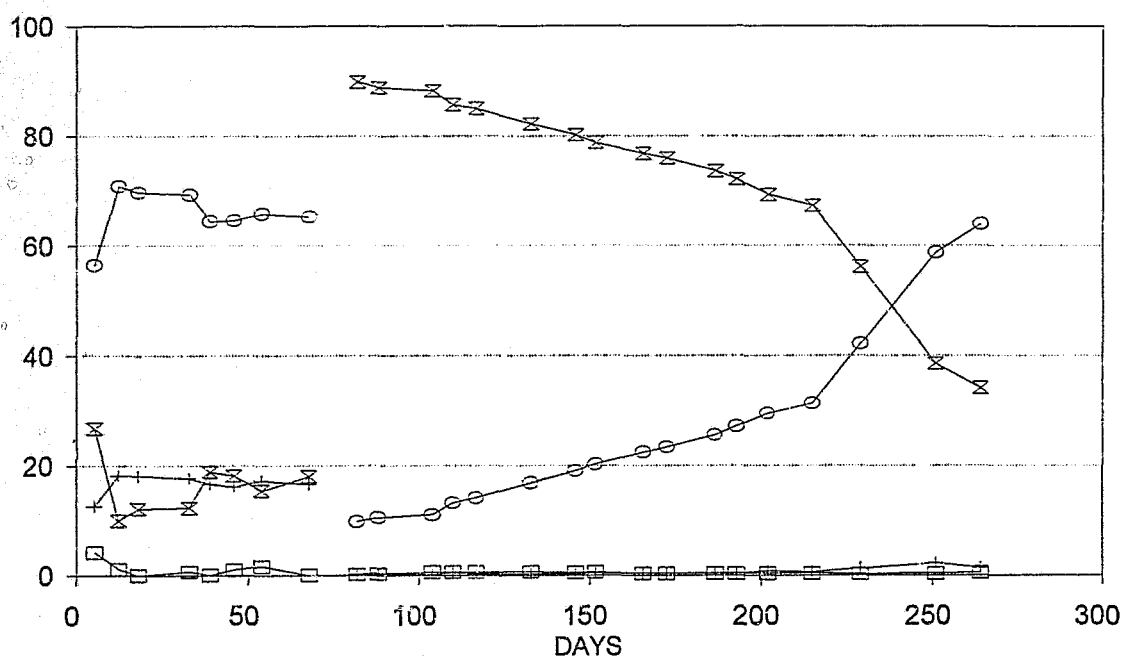


Figure 6.3 Gas analysis for 220 l drum

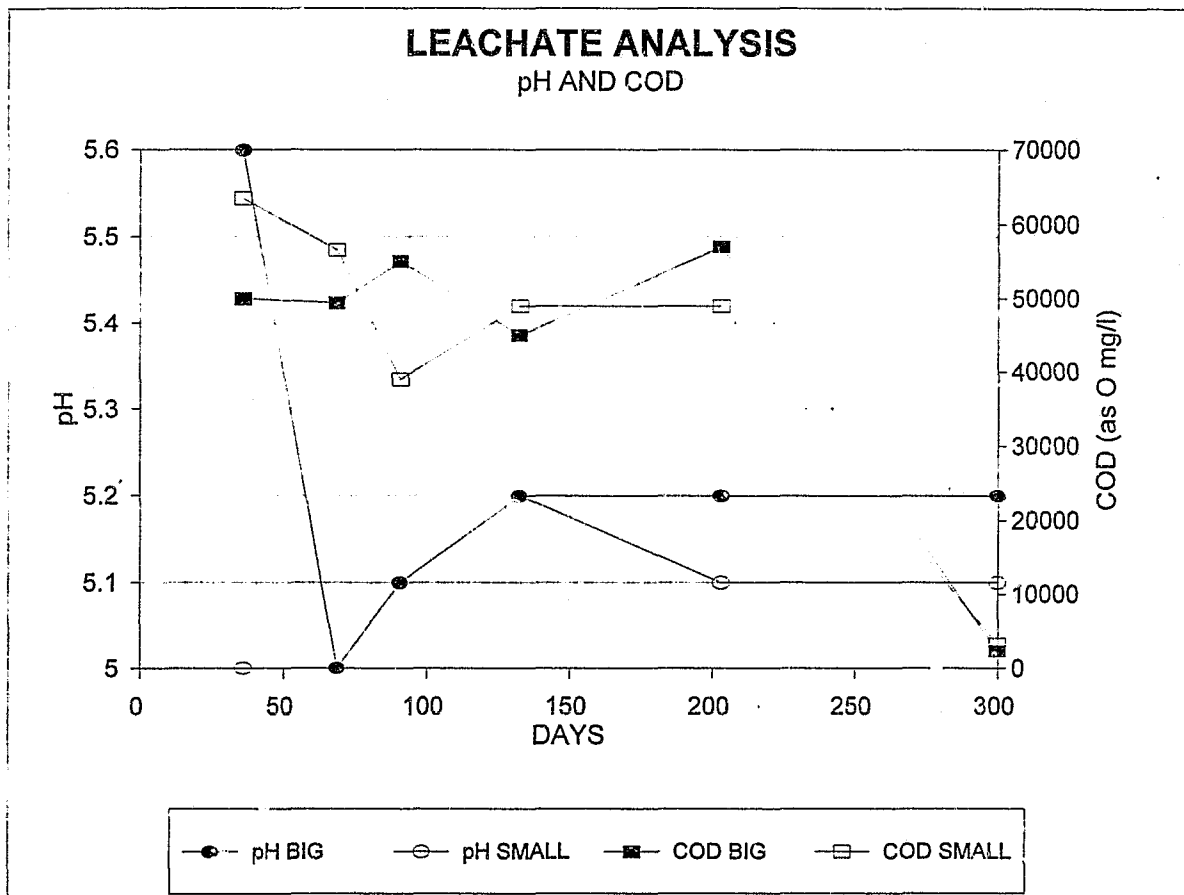


Figure 6.4 Leachate analysis, pH and COD

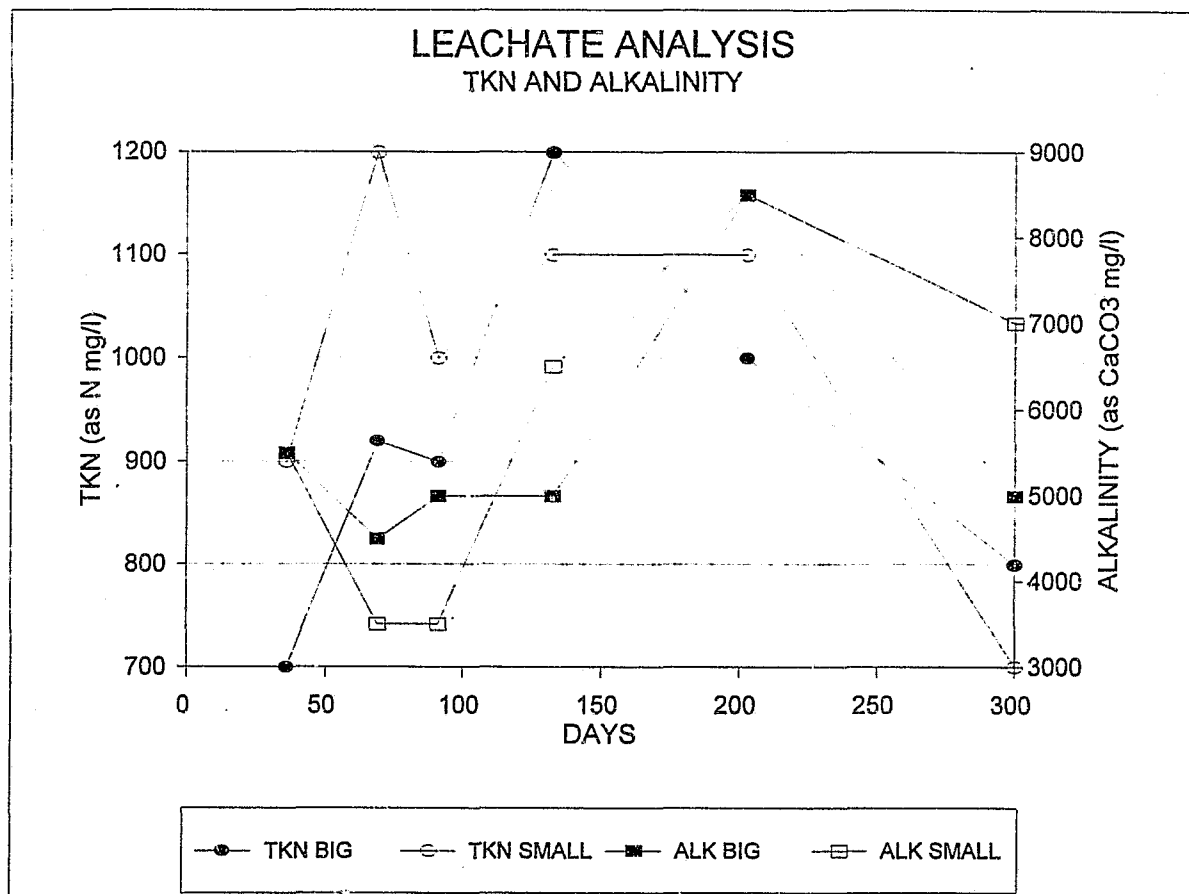


Figure 6.5 Leachate analysis, TKN and alkalinity

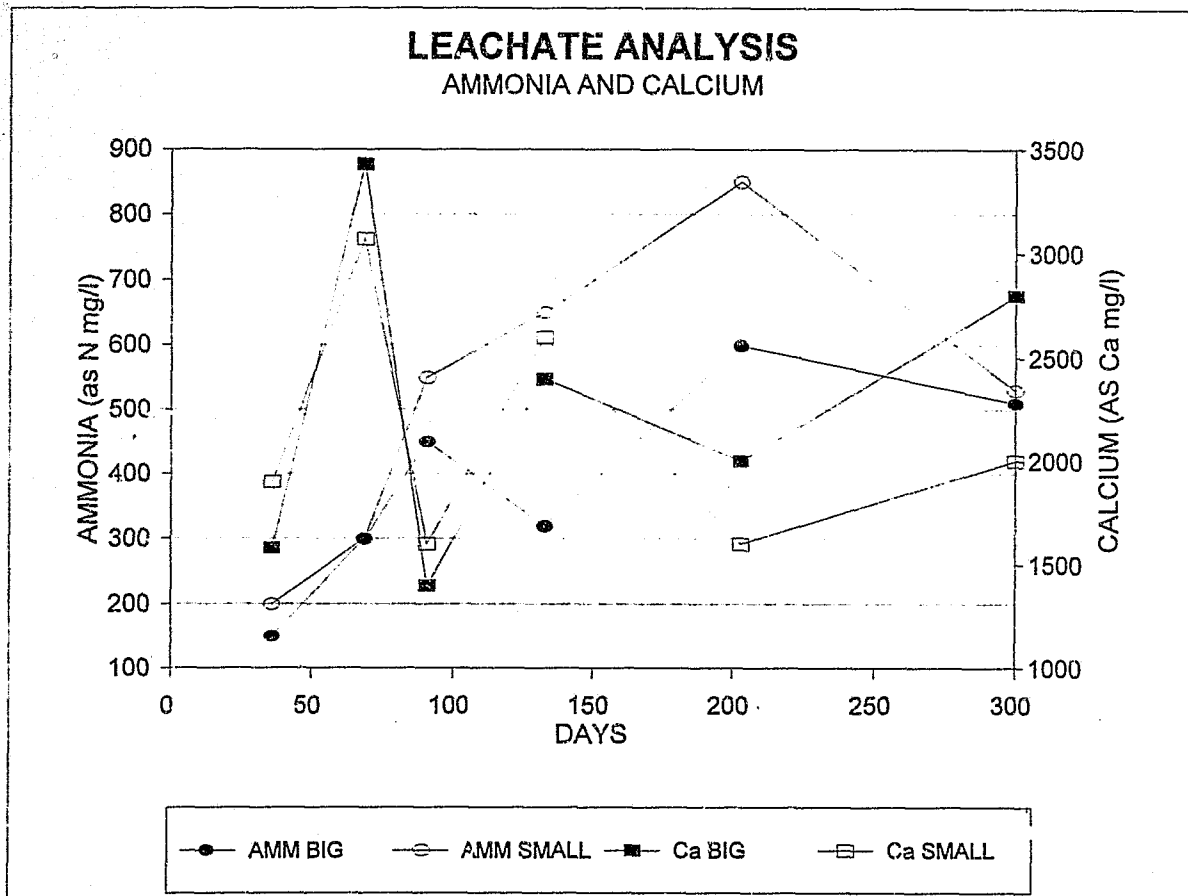


Figure 6.6 Leachate analysis, ammonia and calcium

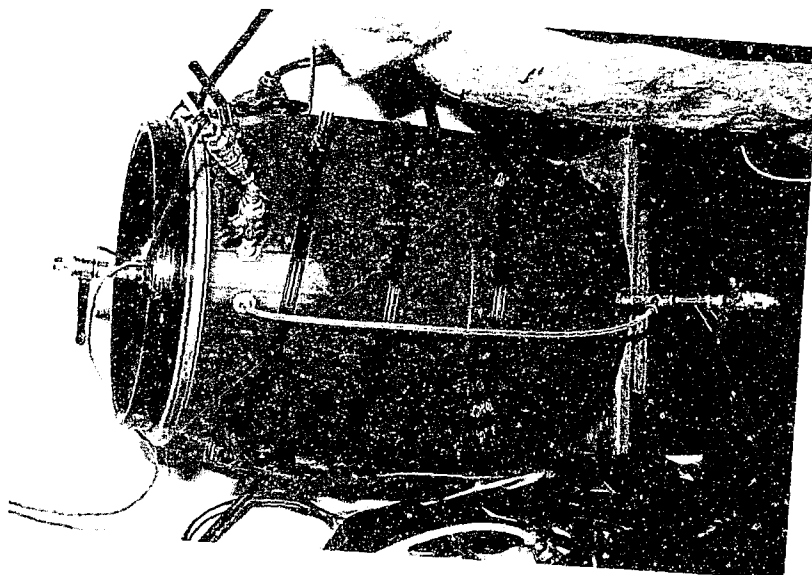


Plate 6.1 50 liter lysimeter showing components

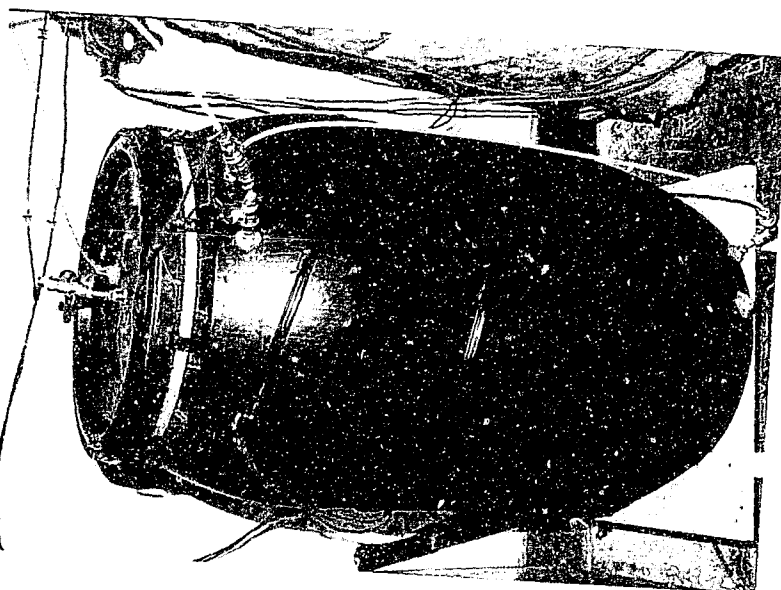


Plate 6.2 200 liter lysimeter showing components

CHAPTER 7 FIELD CELLS

7.1 AIM

In the early stages of the investigation, field cells were identified as an important component of the work to be undertaken. These were to be used, along with the test cells to calibrate the laboratory data with a full scale landfill operation. Large scale field cells provide a means of extrapolating data from the laboratory, where more replicates can be constructed, to the field. Hence more research can be performed at a lower overall cost. Due to their large scale, results become available at a slower rate, hence this component of the investigation was undertaken first.

7.2 STUDY METHODS

7.2.1 Background

Before work on field cells could begin a suitable site had to be selected where the work could continue. After searching and again with the help of the ESC, the new Weltevreden site in Benoni was chosen. The site was chosen mainly due to the separate collection and transportation from two predominantly developed and developing communities. The developed component consisted of waste from Brakpan and the developing component of waste from Watville. Although preliminary studies on waste composition were completed in the Heidelberg area, waste from the Weltevreden site proved to be a good match and was used as the waste source for the lysimeters, test cells and field cells. Classification at the Weltevreden site was conducted when the cells were filled in October 1995, coincidental with another Heidelberg classification so that direct comparisons could be drawn between the two sources of waste.

The Weltevreden site had numerous advantages for the construction of field cells. As with most landfills in Southern Africa, it was in a water deficit area which allowed for the extrapolation of results to other sites. Many studies had been conducted in the East Rand area so that historic generation rates and composition of waste in the area was well understood. There was also space for the experimental cells on the site as it had only recently been commissioned. Lastly, the site was very accessible and there were no personal safety risks. An important consideration when choosing the site was the fact that the site was underlain by an extensive clay layer which obviated the need for a flexible membrane liner in the cells.

In order to facilitate the rapid construction of the field cells, and to create a forum in which any problems could be addressed, the Weltevreden Research Project Work Group was established. This group contained representatives of all the affected parties namely: ESC whose site it was; Wastetech who was the contractor operating the site; Jones and Wagener who were the consultants who designed the site and Wits University who was conducting the research. The membership of the Work Group was as follows:

- ESC - Mr Hennie Neethling
- Wastetech - Mr Jeremy Boswell, Mr Leon Grobbelar, Mr Ryno Gouws
- Jones and Wagener- Mr Danie Brink
- Wits University - Prof G.E. Blight, Prof A.B. Fourie, Mr Jonathan Shamrock

Mr Hennie Neethling was elected chairman of the meetings which were held at regular intervals, both to keep all concerned updated and to act as a forum for ideas and problem solving.

7.2.2 Experimental design

It was envisaged that three field cells were to be constructed. One containing developed community waste, one containing developing community waste and the last one an even mixture of both. To be able to tie the field work in with the laboratory work, it was decided to use a 50% mixture by volume in the mixed cell while keeping note of the mass ratio as well. A record of the total mass of waste going into each cell was also kept. The importance of representability of the waste being placed in the cells was also a vital factor. Waste going into the cells was selected and screened to maintain a fairly constant quality of waste. It was decided to use general domestic waste only and to exclude both dry and wet industrial waste.

Wastetech, being the onsite contractor, agreed to assist in the establishment of the field cells. As the study was to be conducted over the long term, the cells could not be sited anywhere within the active area of the site. It was decided to site the cells south of the existing active phase between the screen berm and the powerlines. This area was put of the foreseen active zone and hence the cells could be part of a long term monitoring program.

At an early meeting it was agreed that there was no need for a flexible membrane liner under the cells and that a compacted, shaped clay base was sufficient. Time was spent at the consultant's office in order to arrive at a practical and cost effective design for the cells. It was decided to change the original base design in order to simplify the system. The base of the cells were sloped in two directions so that leachate converged to one low point. Berms were then constructed on the base to form a basin in which the wastes were placed. A concern was expressed about the effects of biological clogging in the drainage layer design. The Minimum Requirements specifies the stone size in a leachate drainage layer to be between 38 to 50 mm. The point was made that a stone drainage layer would prove to be very expensive. It was therefore decided to use broken

floor slabs from temporary housing that was constructed during site establishment. A tractor, loader, backhoe was to be used to break the slabs into roughly 100 mm blocks which would then act as the drainage layer. It was decided to wet the base after the drainage layer had been placed, both to check if the layer was working and to reduce the chances of leachate permeating the base as the first wetting front came through.

The waste being placed in the cells was to be shaped and compacted by a Rex Trashmaster. The trashmaster would then also be used to cover the cells. The thickness of the cover layers was to be that specified in the Minimum Requirements although it was agreed that the required density values could not practically be met. It was felt that the study would be of more value if it reflected what was happening on the site, hence all compaction and cover operations would be conducted in the same manner as on the rest of the site. To maintain a constant quality of cover over the cells the cover material was excavated and stockpiled before construction began. A concern from the beginning of the project was that the site had previously been used for the storage of mine tailings so that the possibility existed that most of the soil onsite was contaminated by acid mine drainage. To find a source of suitable uncontaminated soil the original geotechnical report was consulted and pH and conductivity tests were conducted on soil samples collected on site. Following a site visit on 7/4/95, samples of the stockpiled cover were taken and pH and conductivity tests were conducted. Following these results further samples, spread throughout the site, were also analysed, see Table 7.1. The samples were prepared and analysed according to standardised soil testing methods (The non-affiliated soil analysis work committee, 1990).

Table 7.1 pH, conductivity and compaction results of soil samples taken at the Weltevreden site

Sample location and description	pH	Conductivity (mS/m)
7/4/95		
Stockpiled cover (ferricrete)	4.0	140
Stockpiled cover (ferricrete)	3.8	130
Wits flowerbeds	7.1	15
13/4/95		
Tailings behind cutoff trench	3.2	100
Freshly excavated ferricrete	3.1	240
Tailings on North West side of site	2.8	550
Virgin soil behind stockpile	5.2	65
21/4/95		
Clay layer in TP1 (0.3 – 1.0 m)	4.0	190
Clay layer in TP3 (0.0 – 0.7 m)	4.3	110
Clay layer in TP4 (0.0 – 1.1 m)	5.0	200
Clay layer in TP5 (0.0 – 0.8 m)	4.8	230
Water samples		
Groundwater in test pit 1	3.5	>1000
Groundwater in test pit 3	3.6	>1000
Groundwater in test pit 5	3.7	>1000
Groundwater in cutoff trench	2.5	>1000
	Optimum moisture %	Max. dry density kg/m³
Test pit 1	12	1950
Test pit 3	11.5	1960
Test pit 5	9.5	2025

From this data it became apparent that all the available cover material on site was contaminated by acid mine drainage. The low pH of the soil would cause a drop in the pH of water passing through it which could have a significant effect on the biological decomposition of the waste, possibly even inhibiting the methanogenic phase. Of all the samples taken on site, the only uncontaminated sample proved to be virgin soil behind the existing stockpile. This source would be used as cover for the field cells. In order to determine if the cover did have a significant effect on the biological decomposition it was decided to construct one additional cell. This cell was a replicate of the 50% mixture but it was covered with mine tailings as was the practice on the rest of the site. See Figures 7.1 to 7.5 for design drawings of the field cells.

7.2.3 Construction of field cells

Following a site visit on the 7/4/95, an area which could be used for the construction of the field cells was chosen. Test pits were dug at the proposed location the following week to establish parameters such as groundwater depth and soil profile. However, due to mechanical breakdown these test pits were only excavated on the 18/4/95. Clay samples were collected from each of the 5 test pits as well as groundwater samples (Table 7.1). From profiling in the holes it was found that the groundwater was on average 1.2 m below the surface with a minimum of 700mm of clay between the groundwater and the proposed waste layer. The clay layer was also found to be contaminated by acid mine drainage, although it was felt that the low pH of the clay was not that important as the wastes were separated from it by a 300mm drainage layer. Compaction tests were performed on the clay samples to determine optimum moisture content and corresponding maximum dry density (Table 7.1). This information was used when constructing the clay layer on site as well as for permeability tests in the laboratory. The clay samples were compacted in a Proctor mould and permeability and acid leach test performed on them.

Excavations for the field cells were started on the 15/5/95. It was decided to excavate into the clay layer in order to get significant slopes on the bases. However, at the lowest point of the excavation, in the south west corner, a perched water table was intercepted. At the time of excavation, it was felt that the water table was deeper, even the test pits seemed to indicate this. The clay was allowed to dry for a week and an attempt was then made to compact the base with a three ton vibratory compactor. The clay base compacted well but as soon as the compactor reached the wet area, it brought up the water again and the machine could not compact. In an attempt to drain the water off a trench was dug all around the perimeter of the excavation, see Figure 7.6 for site layout drawing. The trench was then filled with car tyres which had been cut into quarters, which acted as a cut off trench for the water. A large sump was constructed at the lowest point of the excavation in order to assist with the drainage. After three days the sump, volume approximately 9 m^3 , filled up with groundwater. It was decided to pump out the sump on a weekly basis in order to induce a flow gradient and lower the perched water table. While this was underway, construction of the side berms started. It was decided that once the base was dry, new clay would be imported and the base recompacted. In order to get the compactor in again and to put in the leachate drainage pipes, the lowest side berm was only half built and was completed when work on the base was finally complete. The sump was pumped out on 4 occasions with a noticeable lowering of the groundwater.

In order to drain the area faster a trench was dug from the sump to an area downstream where the water daylighted. This drained the area sufficiently that the imported clay material for the bases could be brought in, levelled and compacted. This was again done by a three ton vibratory compactor with the assistance of a water cart to add moisture. In order to verify the permeability, a study was conducted on the bases using two in-situ methods of liner testing, namely the Guelph permeameter and double ring infiltrometer. Both tests yielded similar results of 1×10^{-7} to $1 \times 10^{-8} \text{ m/s}$. This proved

that the layer could act as a barrier to leachate movement which would channel the leachate to the low point of the cell for collection.

The original idea of using broken concrete slab as a drainage layer was abandoned as it was felt that it could affect the pH of the leachate due to the basic nature of the material. Large quantities of car tyres were available on site and it was decided to use packed tyres as the cell's drainage layer. The drainage layer would channel the leachate to the cell's low point where the leachate drainage sump was constructed. When the drainage pipe was put in it was buried in a trench approximately 200mm below the floor level of the cells. A large sump 1.5 x 1.5 x 0.2 m was excavated, compacted and lined with a screed of 50mm concrete. Curb stones were used to form a head wall into which the drainage pipe was concreted in. A geomesh was used to stop large particles getting into the drainage pipes and blocking them. Once this was completed, each sump was tested by means of adding water until the sump was full and allowing it to drain away.

Before manholes could be constructed at the downstream end of the drainage pipe, a period of wet weather occurred. This weather proved that the drainage layer worked, until the manhole excavations silted up. It also flushed the clay drainage layer and we felt confident that the drainage layer would not affect the pH of the leachate. Experiments conducted in the soils laboratory proved this assumption as once the initial flush went over the clay and leached the contaminants out, the next flushes did not alter pH significantly.

7.2.4 Filling of Field Cells

When the bases had dried out sufficiently the cells were filled with refuse. This took place during September to November 1995. The reason it took 3 months to fill the cells was due to the availability of waste. The rich waste source was in plentiful supply (Brakpan and Benoni) as an average of 30 19m³ compactors visited the site everyday.

The only source of poor waste however was a single compactor per day from Watville. Attempts were made to divert trucks from the transfer station in Khatlehong. When the longhaul compactor containers arrived and discharged their loads the waste was found to be unsuitable due to contamination with rich waste from surrounding areas. Attempts to get uncontaminated poor waste from Khatlehong only proved to be fruitless as the waste streams were mixed at the transfer stations.

During the times that the cells were being filled intermittent periods of rainy weather occurred. The waste was covered at the end of every day with a heavy duty tarpaulin to keep the waste dry. The tarpaulin was removed in the morning and the next loads of waste applied. The cells were filled in the following order;

- Cell 1 - Rich waste only
- Cell 2 - Poor waste only
- Cell 3 - Mixture by volume rich and poor
- Cell 4 - Mixture by volume rich and poor, to be covered with mine tailings.

Each rainy episode filled up the area in which the manholes were meant to be which resulted in the excavations repeatedly silting up. Repeated requests to dig a cutoff trench to divert the storm water could not be met due to the lack of availability of plant on site. Once the cells were filled near the end of November 1995 they were covered and compacted. The berms for the construction of the second lift were then put in place. As soon as the berms were completed a period of excessive rains occurred in which no plant or vehicles could get anywhere near the cells.

When the cells were again accessible a number of problems became apparent. The berms for the second lift had caused all the rain to become ponded above the cells, in places more than 200 mm deep. The manhole ends of the drainage pipes were completely silted up as the cutoff trench had still not been constructed.

7.3 RESULTS

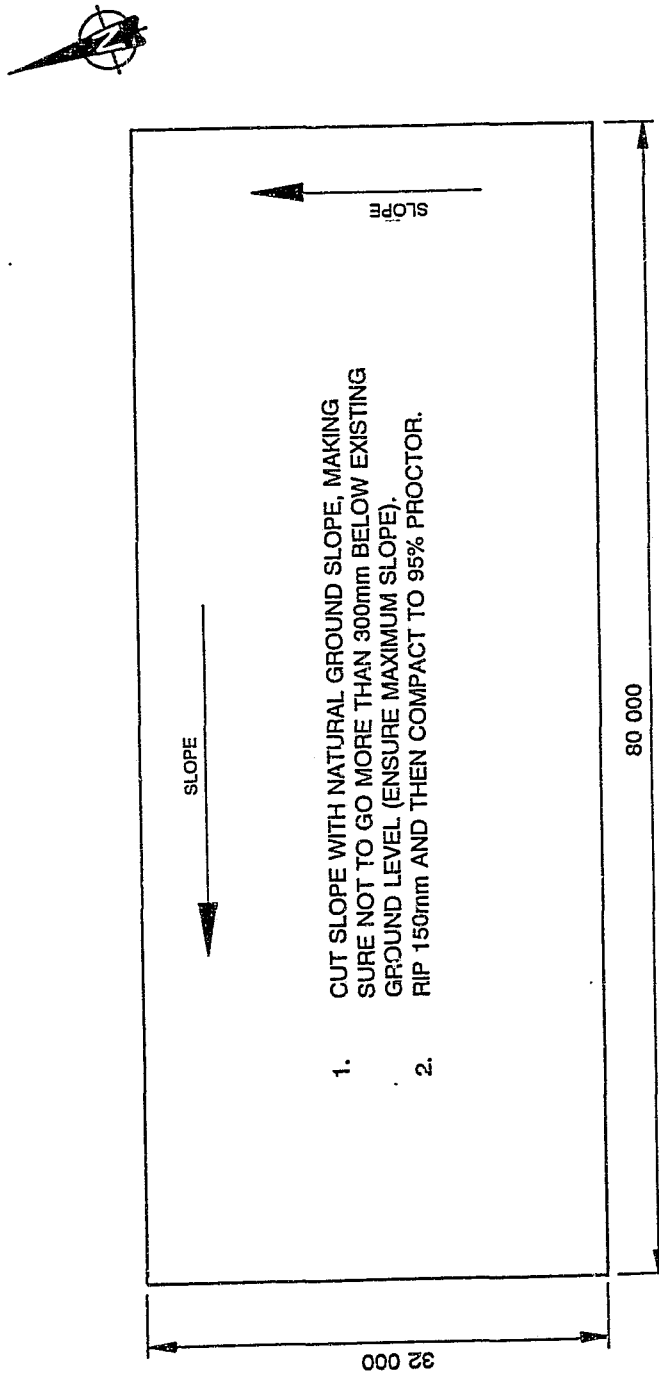
Due to the above reasons it was decided in January 1996 to abandon the field cell investigation as part of this dissertation, but the cells will be used as a basis for future work by a PhD student at Wits University. The main problem was that no leachate data could be used as it was not representative of real landfill conditions. The idea of slope management which directs runoff from the top of the landfill and which is used in the operation of all sanitary landfills is in stark contrast to our field situation in which a large degree of ponding was allowed to occur thus producing totally unrealistic leachate volumes. In order to get a window into the black box of decomposition processes, time series data is needed as well as volumes so that pollutant loads for specific periods can be calculated. Due to the sumps being silted up all the leachate for the entire commissioning period was dammed up inside the cells.

The full monitoring program for the field cell studies was thus abandoned, this included water balance calculations, rainfall and evaporation measurements and possible gas extraction and analysis and the placement of benchmarks to monitor settlement.

7.4 CONCLUSIONS

Valuable lessons were learnt during the period in which this work was completed, reinforcing the classification data and providing an insight into landfill operations. Unfortunately no data from the field cells could be used, although the site did provide a further refuse classification and served as a source of refuse for the test cells and lysimeters which will be discussed in the next sections.

STAGE 1 - BASE PREPARATION



Scale - 1 : 25

Figure 7.1 Base preparation for test cells

STAGE 2 - CELL PREPARATION

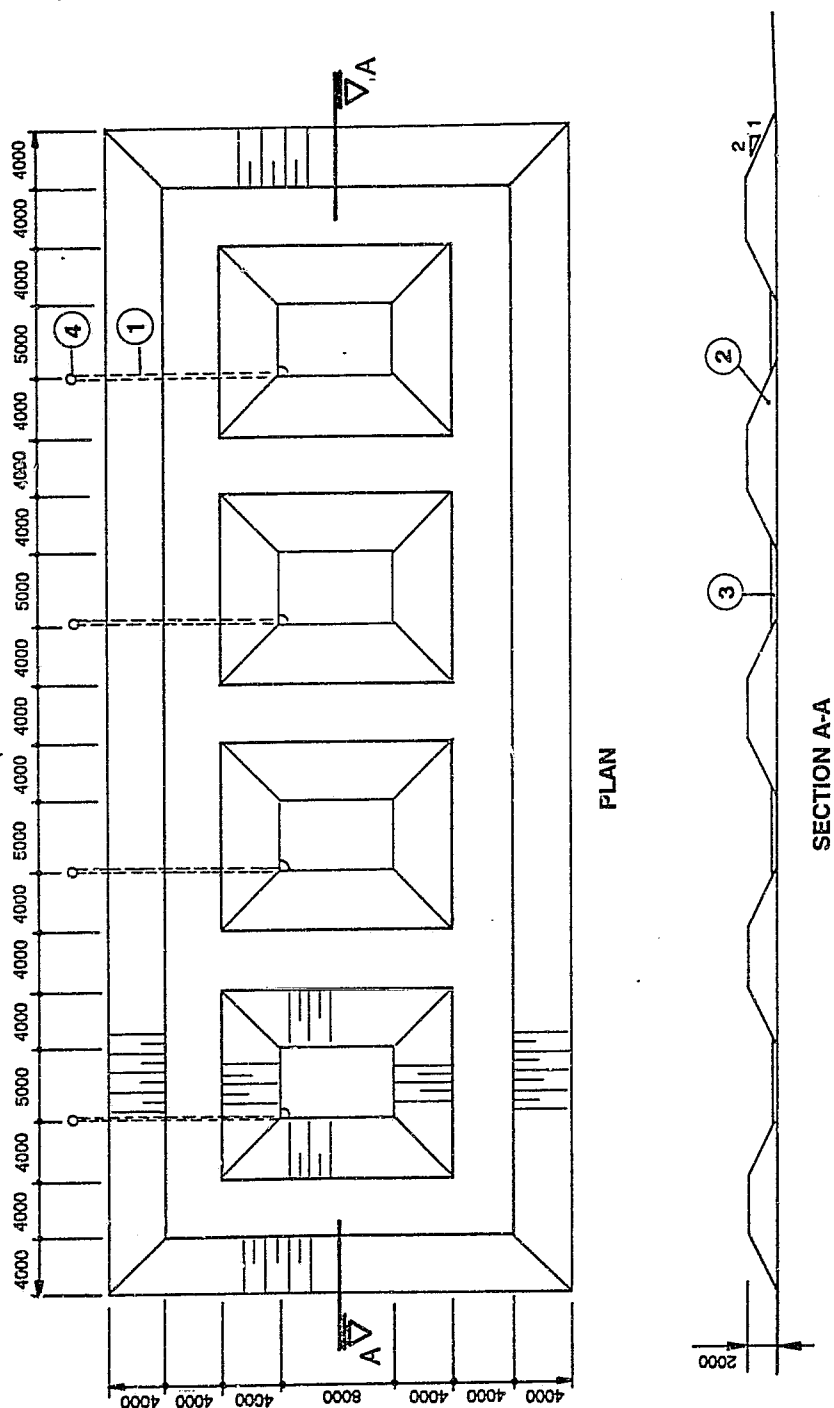
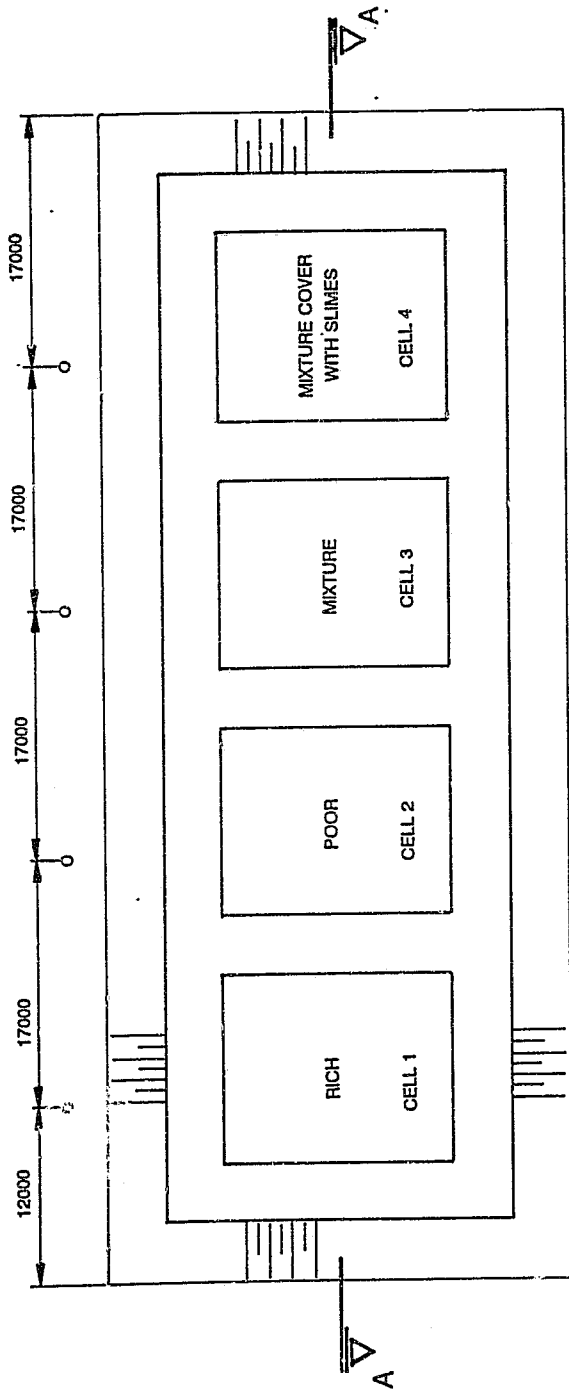


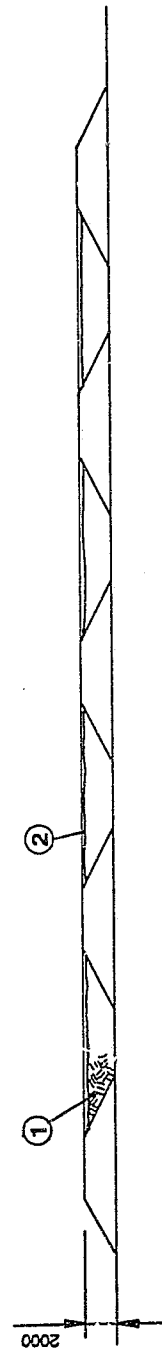
Figure 7.2 Cell preparation for test cells

1. Dig shallow trench for leachate drainage pipes, slope 1:14, install 15m HDPE pipe in each.
2. Construct berms with 1:2 side slopes.
3. Place drainage layer on bottom of cells and at far up the slope as possible.
4. Construct cultrap and then collection manholes from Rocla manhole rings.

STAGE 3 - FILLING OF CELLS



PLAN

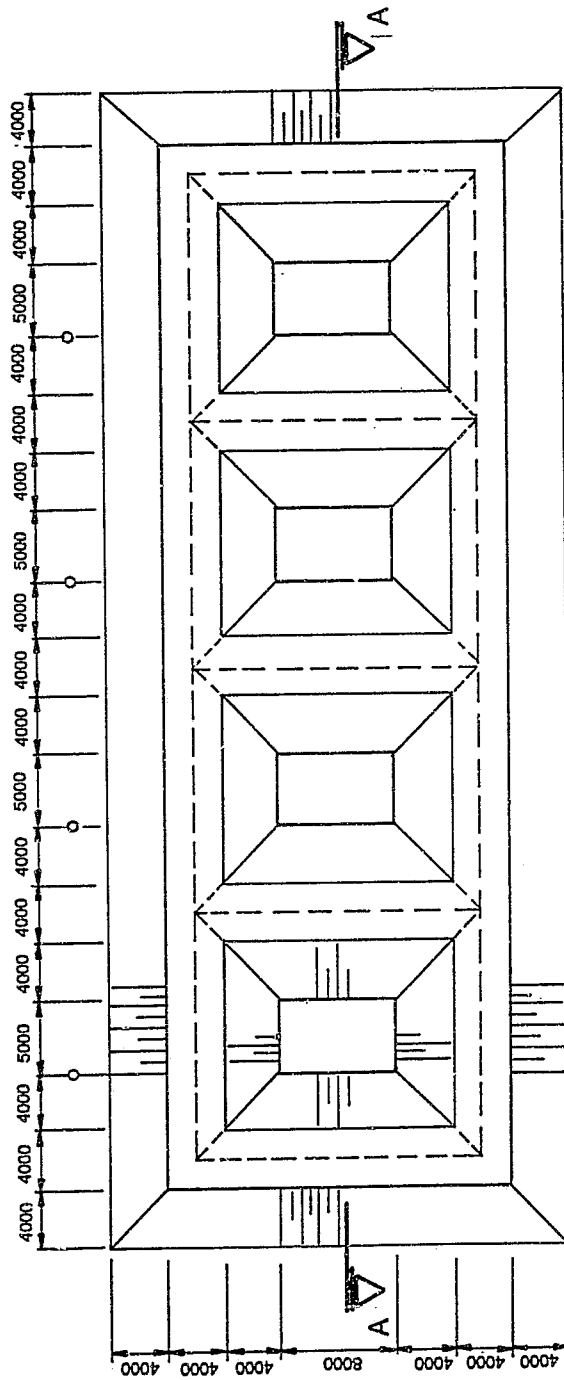


SECTION A-A

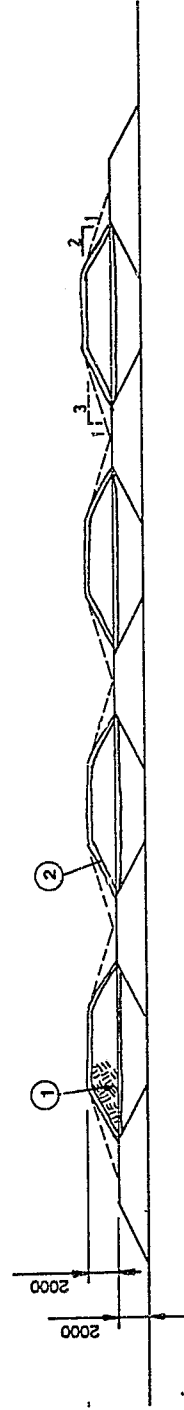
Figure 7.3 Filling of test cells

Volume of first lift
250m³ in each cell

STAGE 4 - FINAL LIFT



PLAN



SECTION A - A

Volume in second lift
250m³ in each cell.

Figure 7.4 Final lift for test cells

1. Place and compact second layer of refuse. Attempt final slope of 1:2. If that is to steep then use 1:3 (dotted line).
2. Cover cell with final cover (call 4 with slopes) 300mm thick.

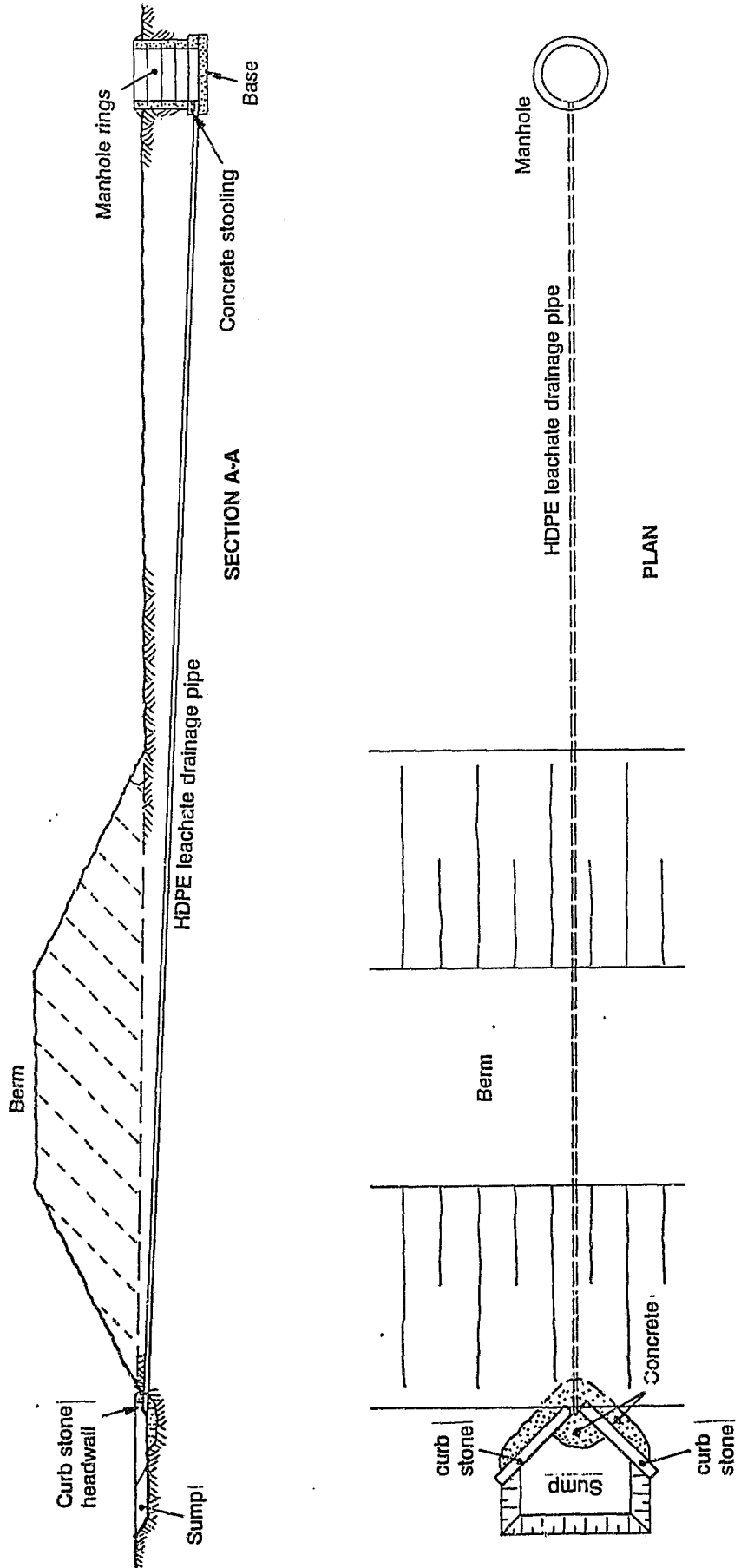
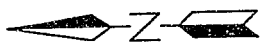


Figure 7.5 Detail of leachate drainage system

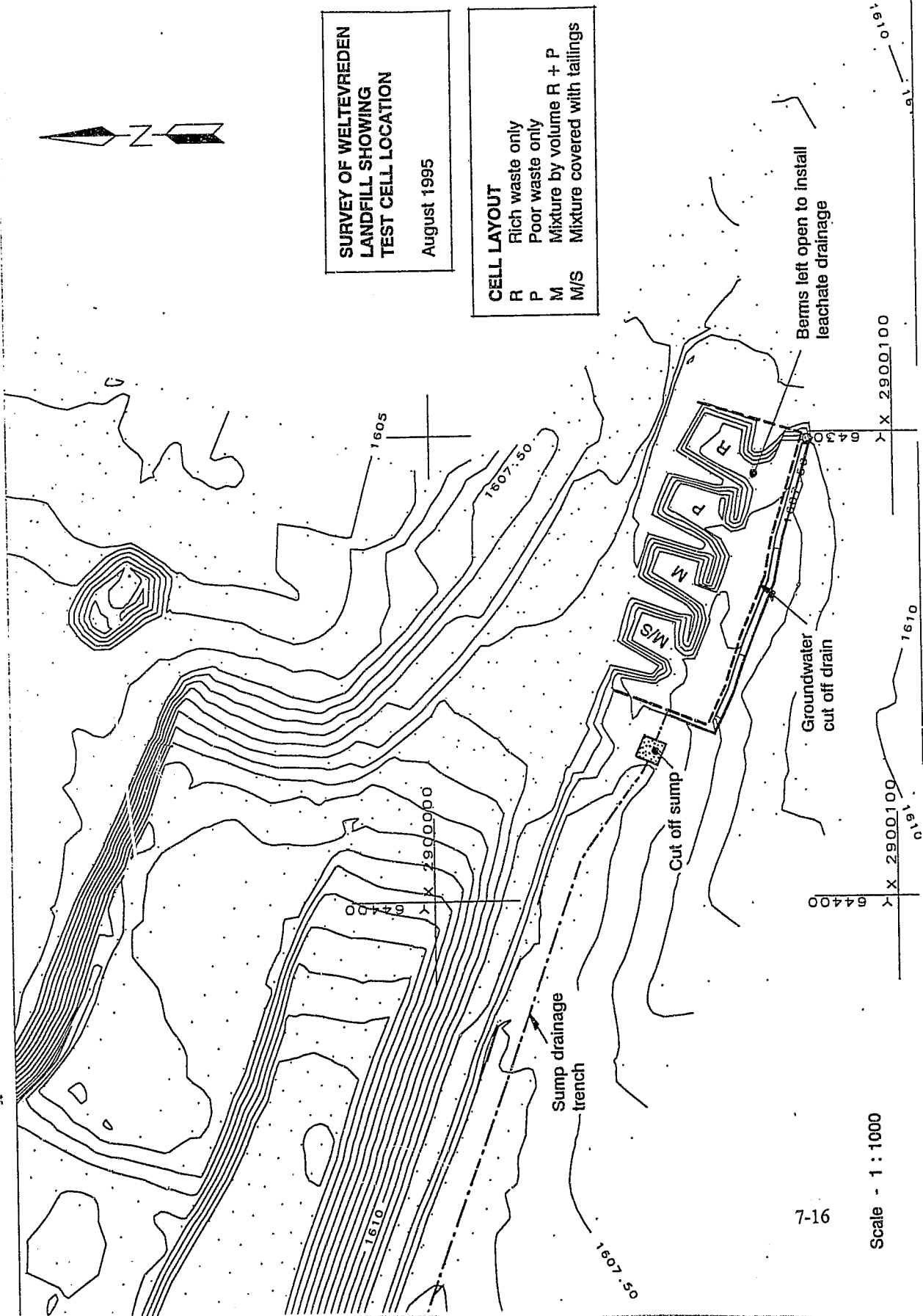


**SURVEY OF WELTEVREDEN
LANDFILL SHOWING
TEST CELL LOCATION**

August 1995

CELL LAYOUT

R Rich waste only
P Poor waste only
M Mixture by volume R + P
M/S Mixture covered with tailings



7-16

Scale - 1 : 1000



Plate 7.1 Excavation of field cell area

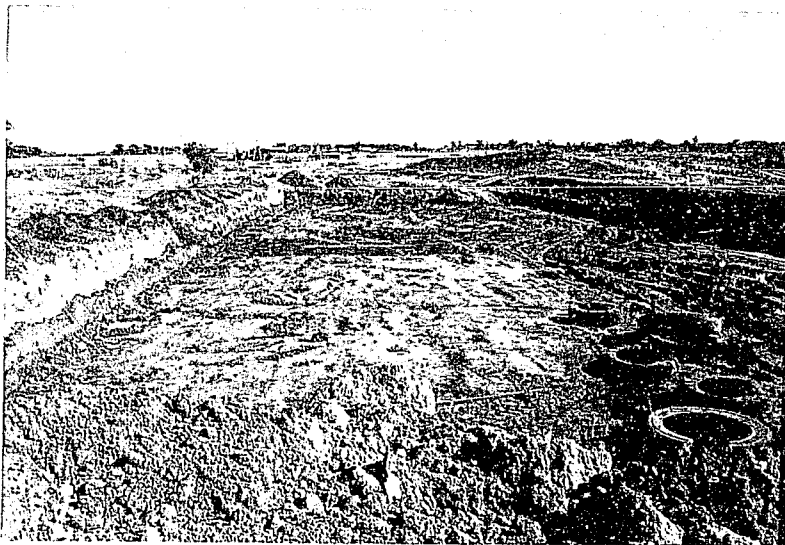


Plate 7.2 Shaped base showing cutoff trench

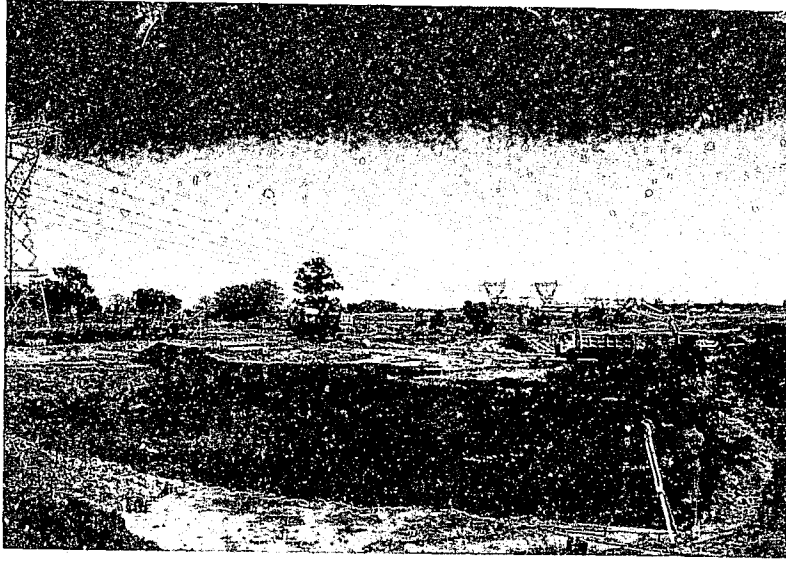


Plate 7.1 Excavation of field cell area

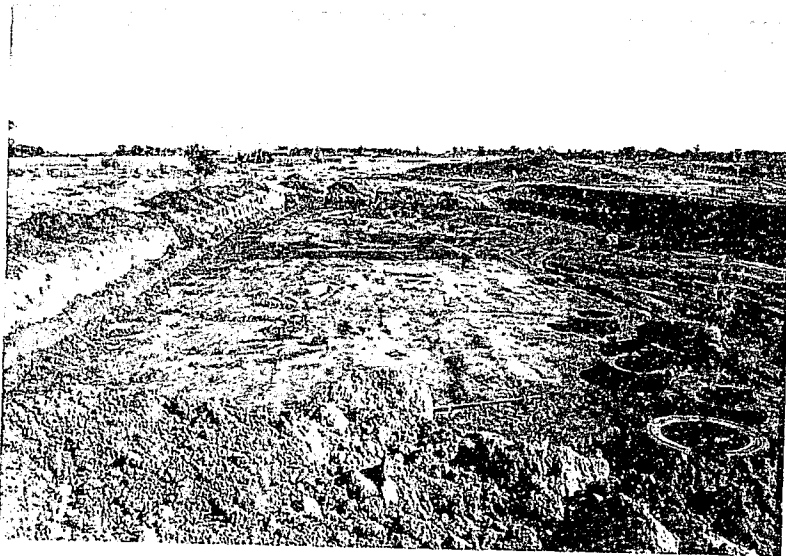


Plate 7.2 Shaped base showing cutoff trench

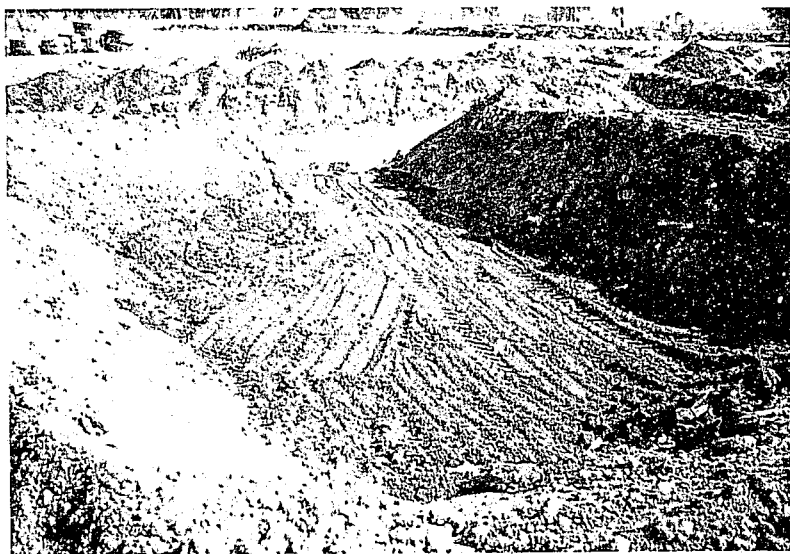


Plate 7.3 Imported clay liner in cell

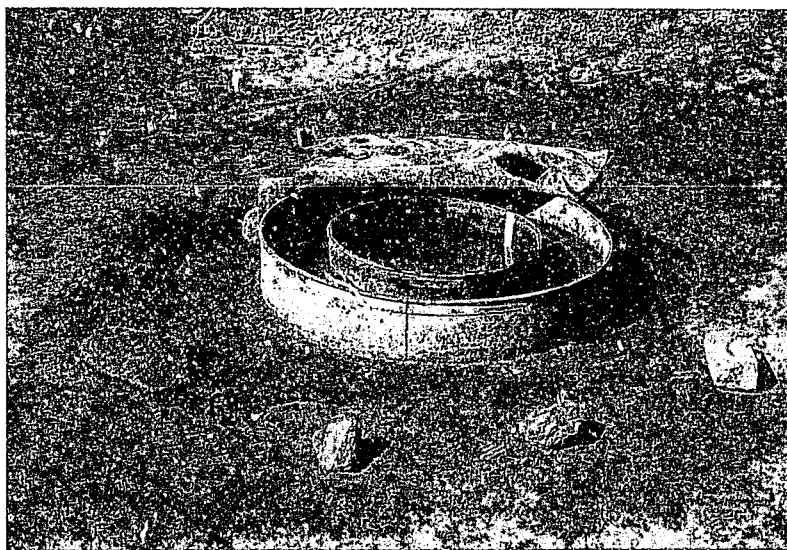


Plate 7.4 Double ring infiltrometer test on clay base

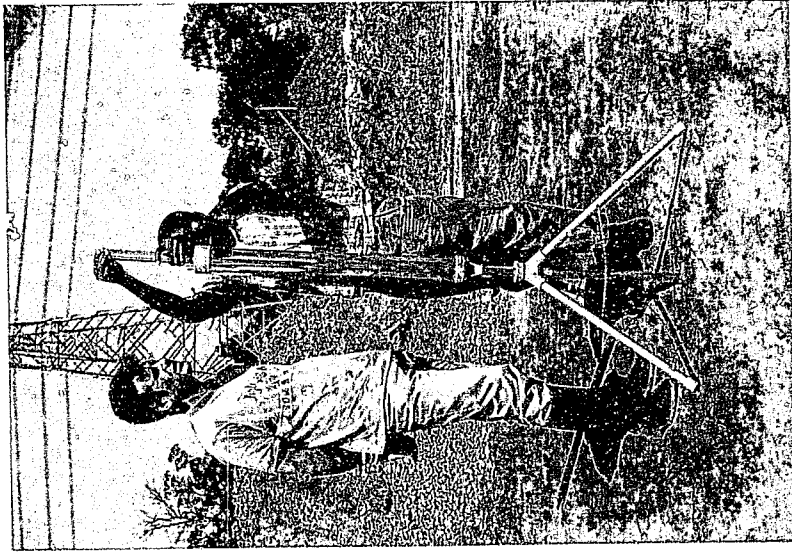


Plate 7.5 Guelph permeameter tests on clay base

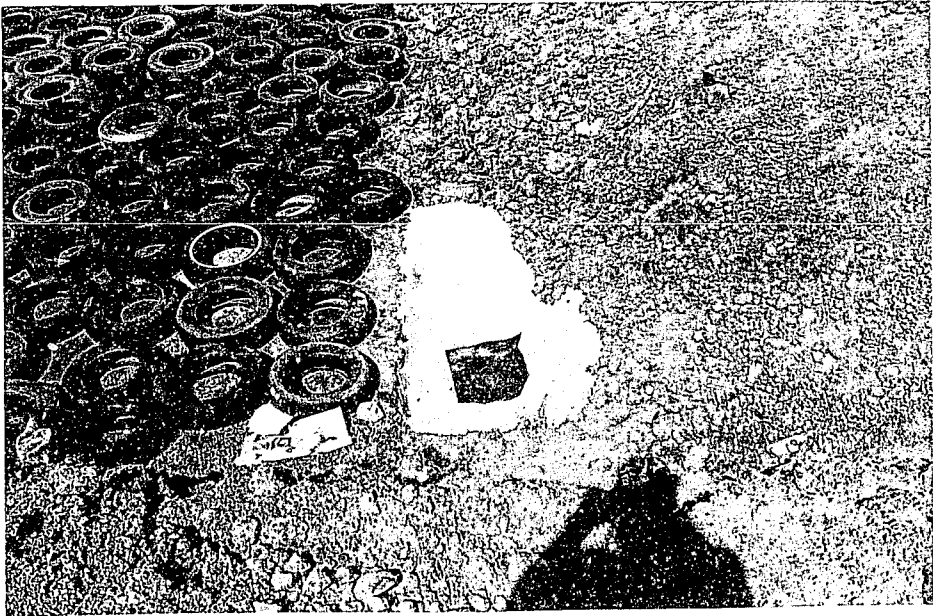


Plate 7.6 Completed sump and drainage media



Plate 7.7 Compaction and covering of refuse

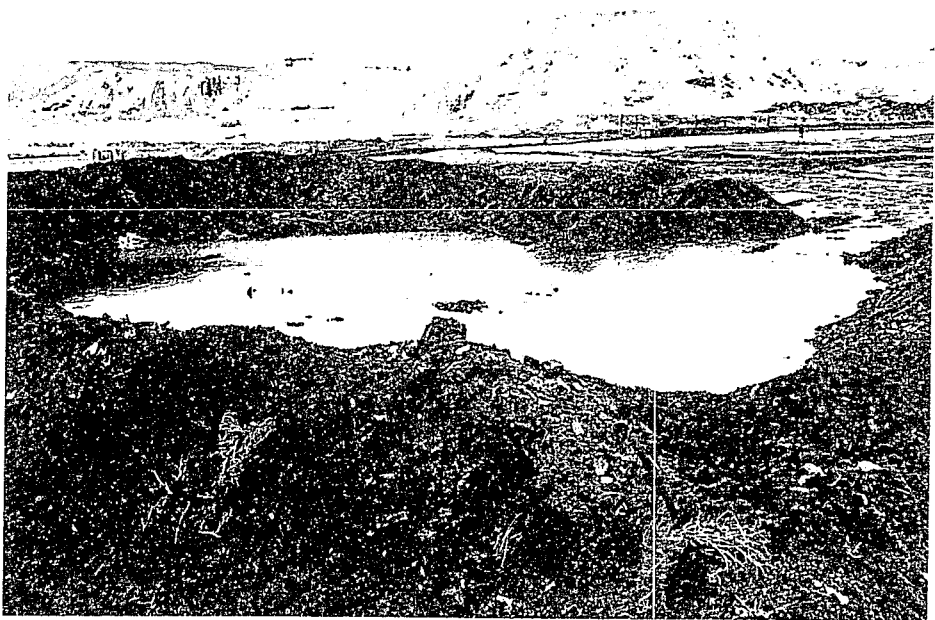


Plate 7.8 Flooding of test cells due to excessive rains

8.1 AIM

In order to study refuse decomposition on a large scale, but still under the direct control of the Department, two test cell lysimeters were constructed. It was felt that the large scale, roughly 3.5 m^3 , could act as a link for extrapolation of data from the lysimeter studies through to the full scale field cells. As the field cells had to be abandoned it proved to be fortuitous as the test cells turned out to be the only source of large scale decomposition data.

8.2 STUDY METHODS

8.2.1 Test cell construction

The test cells were constructed of three vertical double brick walls on the western wall of the Civil Engineering building. Two sloping concrete floors were cast sloping towards the open end. Treated wooden panels were used to seal off the front of the cells. This was done to allow for the removal of the waste once the experiment is completed. The resulting rectangular containers had a volume of approximately 3.5 m^3 , see Figure 8.1. In order to prevent excessive ingress of rainwater into the cells, gutters were constructed above the cells to divert runoff occurring off the side of the building. The two cells were lined with a 2 mm thick geomembrane cut and folded in such a way that a continuous basin was formed inside the cells. The only opening excluding the top was one small hole at the lowest point which was used as the leachate drainage point. A piece of threaded 2 inch pipe with fixing nuts, pressure washers and sealing washers was used to connect the inside of the cell through the geomembrane and wooden

partition to the outside. A 2 inch gate valve was attached to the pipe in order to seal it off. All joints when constructed were sealed with silicon sealer.

In order to get the leachate to the drainage orifice it was necessary to construct a leachate drainage layer. The layer was formed from 22 mm crushed andersite aggregate, formed in a layer 100 mm thick. Before the stone was placed in the cell it was washed with water to remove all the dust and crushing residue. A concern was expressed that the layer would alter the chemical composition, specifically pH, of the leachate as it percolated through it. In order to test this theory, tests were conducted in which samples of the stone were placed in an acid solution of known pH as well as a distilled water solution of known pH. pH readings were taken again at 12, 24 and 48 hours after initial emersion. No change in either pH was observed. In order to prevent the stones getting into the leachate drainage pipe and gate valve, a small piece of geogrid was placed over the orifice before the stone layer was constructed. When the drainage layer was in place tests were conducted firstly to see if the drainage layer worked and secondly to see if the leachate drainage pipe, fittings and gate valve sealed. A leak was detected in the left-hand cell between the drainage pipe and the geomembrane. The leak was repaired, cell tested again and found to be in good working order.

8.2.2 Test cell filling

The test cells were filled on the 3rd to the 6th of November 1995, one with rich waste and one with poor. The refuse used was the same as that used to fill the field cells i.e. rich waste from Benoni-Brakpan and poor waste from Watville. A three ton truck was used to collect the waste. As the truck entered the site, it was weighed at the weighbridge. The truck was filled using the TLB on site and the truck was weighed again as it left the site. Once the waste was in place on the truck it was covered with a tarpaulin to prevent any moisture addition or loss.

In order to ensure that the refuse conditions were similar at all phases of the investigation it was necessary to compact the refuse. The piece of equipment best suited to this was a Wacker trench compactor. As the weight of refuse was known from the weighbridge slips and the volume of the cell was known it was possible to work out the achieved densities of compacted waste. The compactor worked well as we were able to compact the poor waste to 807 kg/m^3 and the rich waste to 587 kg/m^3 . Careful note was taken to utilise the same amount of compactive effort on each cell i.e. the same number of passes were used per compacted layer. Each compacted layer measured approximately 300 mm in depth. The waste's moisture content at placement was 67% by dry weight for the rich waste and 38% by dry weight for the poor.

Due to the location of the cells we were extremely concerned for the potential for volatile gases and odours. It was decided to use a blend of 50% compost and 50% topsoil to cover the cells. The refuse was covered with a 200 mm thick layer of this mixture and the cover layer was again compacted on each cell using the Wacker. pH and conductivity tests were conducted on this cover and it was found to be neutral. During the course of the experiment a substantial growth of vegetation appeared on top of the test cells due to the organic rich nature of the topsoil and compost. This vegetation growth was not removed and was left to grow.

8.2.3 Test cell monitoring

The monitoring program for the test cells set up at the beginning of the investigation was as follows:

1. Precipitation

Initially there was only one raingauge placed in the middle of the two cells. It was decided about a month into the investigation to include a second gauge in the cells as well as a gauge on the roof of the Civil Engineering building. These gauges were read

and emptied every week day morning. The extra gauges proved only to confirm the readings of the original gauge. The rainfall value for any day was taken as an average of the three gauges.

2. Leachate

Leachate was sampled every two weeks from the date of the first leachate being detected. The drainage layer was completely drained on each occasion. The samples were sent to the Johannesburg City Council's CYDNA Laboratories for analysis.

3. Temperature

Thermocouples (copper constantan with a plastic insulating sleeve) were driven into the centre of each waste mass. They were then used, along with a thermocouple bridge, to measure the internal temperature of the waste every two days.

4. Gas

It was envisaged that samples of the gasses produced could be taken and analysed every two weeks. In order to achieve this a gas sampling apparatus was constructed and tested. This consisted of a two litre cold drink bottle of which the bottom was removed. A piece of tubing was inserted into the lid and sealed in place. A hole was made in the soil cover and the bottle's open end placed in the hole in contact with the waste body. The cover was reinstated around the bottle and recompactd so that any gaseous emissions given off by the waste could be sampled through the tube in the lid.

8.3 RESULTS

The experiment was monitored for a period of 721 days for this dissertation. The results will be presented in the order of water balance, leachate, leachate pollutant loads, temperature and gas.

8.3.1 Water balance

Rainfall data was entered as and when it occurred. The rainfall in mm was taken as the average between the two raingauges on the cells and the raingauge on the roof. To convert the rainfall to a volume of water entering the cell, the average gauge reading in mm was multiplied by the top surface area of one cell and converted to litres. The same value of water input was taken for the rich and poor waste. Runoff from the cells was limited so that the majority of rainwater falling on the cell was allowed to infiltrate.

Each time a leachate sample was taken all the leachate in the drainage layer was allowed to drain out. A leachate sample used for quality determination was usually taken half way through the procedure. Once all the leachate had been extracted the bucket was weighed in order to determine the volume of leachate in litres. A cumulative water balance was conducted for the two cells. See Table 8.1 at end of chapter and Figure 8.2.

The initial moisture content of the wastes was 67% for the rich waste and 38% for the poor waste by dry weight. This can be converted to a mass of water in each cell by using the following equation:

$$m_w = \frac{m_{tot}}{(1 + \frac{1}{m_c})} \quad \text{Craig 1987}$$

where m_w = mass of water in each cell
 m_{tot} = total mass of refuse in cell
 m_c = moisture content of refuse

The rich waste had a total initial mass of water of 514 kg while the poor waste had 485 kg of water, only 29 kg difference. When the cells started producing leachate however,

the poor waste produced consistently more leachate than the rich, only reaching the same value at day 112.

This could be a function of both the field capacities of the waste as well as how available the moisture within the waste is to leaching. The moisture content tests were conducted by oven drying three samples of the two wastes over a period of two months. These samples were dried at 50°C until no change in mass was observed, usually around four days, and taking the average of the results. The majority of the water in the rich waste is in the organic matter and only becomes available through decomposition. Due to the high paper content in the waste it also initially absorbs more liquid to satisfy the field capacity. The poor waste conversely had only to satisfy the field capacity of the majority ash content, after which any addition of water would produce leachate.

Around day 350, the system had stabilised with the leachate produced amounting to 27% of the total input by rain. If we examine the water balance equation the following deductions can be made:

$$L = (P + W_m + B + L_i) - (R + E_t + G) \pm \Delta S$$

where

L = leachate (known)

P = precipitation (known)

W_m = initial moisture content (known)

B = water produced due to biological action (assumed negligible)

L_i = lateral inflow (known to be zero)

R = runoff (known to be zero)

E_t = evapotranspiration (unknown)

G = vapour loss in gas (assumed negligible)

ΔS = change in storage (unknown)

The only unknowns in this equation are evapotranspiration and the change in storage or field capacity. As the cells were producing leachate we know that the storage capacity had been reached, evident in leachate production by day 21. Thus by far the largest form of loss of water was by evapotranspiration. This figure amounts to an average loss up to day 351 of 1770 litres or 576 mm. If we examine the curve after day 351 which is the point at which the rainy season begins there is a time lag until day 440 when leachate production again begins. This indicates that the cells have to absorb approximately 1500 litres before leachate production started to occur again. This seems to indicate that the waste body had dried out over the winter period. This is consistent with findings reported by Roussev (1995), who found that in the semi-arid climate of Johannesburg, drying of waste occurs up to depths of up to 5 m. By the end of the monitoring period the cumulative volume of rainfall was 1369.1 litres for the rich waste and 1062.7 litres for the poor waste. The rich waste thus produced leachate amounting to 24.7 % of rainwater while the poor waste had produced leachate amounting to 19.2 % of rainwater.

8.3.2 Leachate quality

8.3.2.1 Introduction

Leachate samples were taken roughly every two weeks. The drainage layer was totally drained with the sample for analysis taken in the middle of emptying each cell. The cell was drained by using a length of hose attached to the gate valve outlet and placing the collection bucket downhill of the cells. This ensured that no air was allowed into the drainage layer during leachate removal. The samples were then placed into glass Schott bottles and sent to the Johannesburg Council's CYDNA laboratories for analysis. Due to cost restraints the full range of water quality indicators could not be tested for. Instead a list of 9 key parameters were chosen from literature and experience. These

were pH, COD, alkalinity, conductivity, chlorides, ammonia, nitrate, nickel and potassium. See Tables 8.2 and 8.3 at the end of the chapter.

8.3.2.2 Leachate pH

See Figure 8.3. The pH of samples was initially done by CYDNA laboratories but this function was later taken over by the Department of Civil Engineering with the establishment of a water quality laboratory. Initially, up to day 100, the poor waste's pH was consistently higher. This trend however, reversed and thereafter both the poor and rich wastes' pH oscillated around the same values. The only broad trend visible from the data is a change from general acidic conditions, indicative of acetogenic decomposition, to general alkaline conditions when methanogenic bacteria would be expected to be predominant. This changeover seemed to occur around day 100. Another outstanding feature of the curve is a sharp rise in pH around day 239.

8.3.2.3 Leachate COD

See Figure 8.4. The Chemical Oxygen Demand shows a broad decreasing trend with time. This curve is evident in the literature and was expected. What was not expected however, was that initially the poor waste COD values were considerably higher than those of the rich waste. The values reach parity around day 98 and continue to fall at the same rate in the shape of an exponential curve, indicating that as the waste breaks down there is less organic matter available for decomposition.

8.3.2.4 Leachate Alkalinity

See Figure 8.5. The alkalinity showed marked peaks at day 100 and day 300 on a exponentially decreasing curve. Initially the poor waste's alkalinity was higher than the rich waste, which was as expected due to the presence of ash in the poor waste.

However, after the initial peak of day 100, values tended to normalise and reflect similar trends. It is suspected that the first peak is indicative of the changeover to methanogenic conditions, the second peak at day 300 corresponds with other curves and seems to coincide with a stabilisation in quality parameters.

8.3.2.5 Leachate conductivity

See Figure 8.6. The conductivity, expressed as mS/m, also exhibits a decreasing curve. The conductivity measurement was also taken over by the Department of Civil Engineering. At the start of the experiment, the conductivity of the poor waste was consistently higher than that of the rich waste. Around day 100 again, a change occurred and both values stabilised close to 1500 mS/m. Near day 280, the curve starts to taper off very rapidly although the poor waste conductivity values remained consistently higher than that of the rich waste. The declining curve indicates an improving quality due to refuse stabilisation.

8.3.2.6 Leachate chloride

See Figure 8.7. The chloride curve shows an almost straight line curve with the average value around 100 mg/l apart from a very marked peak around day 100. As chloride values would not be significantly altered by the biochemical changeover from acetogenic to methanogenic breakdown, this marked peak is most likely due to a wash through of contaminants after a period of heavy rains.

8.3.2.7 Leachate ammonia

See Figure 8.8. The ammonia curve also exhibits a significant peak at day 100 and thereafter stabilises. Through the peak and during the subsequent stabilised phase, the rich waste has higher concentrations of ammonia than that of the poor waste. This was

as expected as the ammonia is indicative of recent decomposition of organic material, of which there is more in the rich waste. Around day 300 the concentrations again drop, with the concentrations reaching similar values. This decline continues until hardly any ammonia was being produced in the leachate, again indicative of refuse stabilisation.

8.3.2.8 Leachate nitrates

See Figure 8.9. From the supplied data of nitrate concentrations no conclusions can be drawn. The large sporadic fluctuations in concentrations seem to indicate a problem with the analysis technique.

8.3.2.9 Leachate nickel

See Figure 8.10. The nickel concentrations also show a broadly decreasing trend. Initially the concentrations are high and they reach a minimum around day 100, thereafter they increase and stabilise and around day 300 the curve again increases. After this the curves both decrease. The minimum around day 100 could be due to increased alkalinity and high pH, thus immobilising heavy metals. The high values around day 350 could be due to the wash through effect. One noticeable feature however, is that the rich waste's concentrations are consistently higher than those of the poor waste.

8.3.2.10 Leachate potassium

See Figure 8.11. The potassium concentrations show a definite decreasing curve with the rich waste again being consistently higher. Potassium concentrations do not seem to be influenced as the other parameters as no peaks are evident at day 100 or day 300.

8.3.2.11 Conclusions

The concentrations of contaminants, in general, decreased over the course of the experiment. The number of discrepancies serve to demonstrate the complexities of the biochemical processes that take place with biodegradation of waste and the need to consider the range of parameters when considering the status of the decomposition process. To eliminate the effect of water volume, i.e. considering concentrations it was decided to evaluate the pollution potential of the wastes in terms of pollutant loads.

8.3.3 Leachate contaminant loads

8.3.3.1 Introduction

The pollutant loads were calculated by multiplying the concentrations of a specific contaminant on a day with the value of leachate extracted on that day. The result is a cumulative curve of the mass of a specific contaminant over the course of the experiment. See Table 8.4 and 8.5 at the end of the chapter. For days when quality parameters were not available, due to technical problems at the laboratory, linear interpolation was applied to the data to fill in the missing points. This was done due to the fact that the analysis applied to the data was of a cumulative nature and any gap in the data resulted in a lowering of the cumulative effect. Interpolated data is shown in italics in the tables.

8.3.3.2 COD load

See Figure 8.12. Surprisingly, the COD load for the poor waste is higher than that for the rich waste. This trend is contrary to what was expected and could be as a result of the procedure of obtaining COD's. The test oxidises all the oxidisable components in a sample, whereas the BOD is an indication of those substances that can be oxidised by

bacteria. Both curves have the same general trend to increase sharply in the beginning over a period of high rainfall and then taper off as time progresses, indicating stabilisation of the waste. At the end of the monitoring period, the poor waste had leached 3503.8 g or 32.7% more COD than the rich waste.

8.3.3.3 Alkalinity load

See Figure 8.13. The alkalinity of the poor waste is consistently higher than that for the rich waste until day 550. This trend is as expected due to the high ash content of the poor waste. This would indicate that the poor waste would be buffered to a greater extent than the rich waste and hence would have conditions more suitable to methanogenesis at an earlier stage. One noticeable trend is that the curve starts increasing slowly and then goes into a phase of rapid increase after which it levels out and again increases from day 400 to day 550. The period of day 400 to day 550 corresponds to a period of increased infiltration washing the pollutants out. At the end of the experiment, the poor waste had leached 42.4 g or 0.3 % more alkalinity.

8.3.3.4 Chloride load

See Figure 8.14. The cumulative load curve for rich waste turns out to be higher although in the initial stages the reverse is true. The curve seems to go through the same gradual increase, peak around day 100 and then tapering off and a second peak around day 400. At the end of the test the rich waste had leached 897.2 g or 31.7 % more chloride than the poor.

8.3.3.5 Ammonia load

See Figure 8.15. The ammonia curve follows the same pattern as previously seen. In the initial stages the poor waste has higher values but after the first peak phase the rich

waste gets progressively higher. This trend was as expected as the rich waste had a greater biodegradable content and hence would have more products from organic degradation in it. At the end of the test the rich waste had leached 431.3 g or 64.4% more ammonia than the poor waste.

8.3.3.6 Nitrate load

See Figure 8.16. As explained in the leachate quality results this data set is erroneous and no analysis can be made.

8.3.3.7 Nickel load

See Figure 8.17. This curve differs from all the previous curves as the pattern of cumulative pollutant loads does not vary in the same fashion. Initially the values start out similarly but as the test progresses the rich waste continues to increase and then level off while the poor waste starts to level off almost immediately. A second increase in the curve occurs at the same time as the cells are subjected to increased infiltration. The nickel load curves are different however as the second peak around day 400 is almost as large as the first indicating that this component may not be so dependent on biological action. The curves indicate either a lower nickel load in the poor waste or possibly that the ash component in the poor waste is immobilising the heavy metal contaminants. The increased alkalinity of the poor waste is also a contributing factor as the alkali conditions discourage the mobilisation of heavy metals. At the end of the test the rich waste had leached 0.15 g or 136.4 % more nickel than the poor waste.

8.3.3.8 Potassium loads

See Figure 8.18. This curve again varies from the established curve shape although matching the curve shape for nickel. The rich waste cumulative load is consistently

higher, this was as expected as the rich waste has more organic matter of which potassium is the most common counterion especially for phosphate. At the end of the test rich waste had leached out 1082.2 g or 142.5 % more potassium than poor waste.

8.3.3.9 Conclusions

To summarise the cumulative pollutant loads, see Table 8.6.

Table 8.6 Summarised cumulative pollutant loads

Parameter	Cumulative Load - Rich (g)	Cumulative Load - Poor (g)	Waste with higher load	% difference
COD	10699.33	14203.13	Poor	32.7
Alkalinity	15127.39	15169.77	Poor	0.3
Chloride	3728.62	2831.40	Rich	31.7
Ammonia	1100.81	669.47	Rich	64.4
Nickel	0.26	0.11	Rich	136.4
Potassium	1841.65	759.41	Rich	142.5

An important factor to consider here is that there was 1760 kg of refuse in the poor waste cell and 1280 kg in the rich waste cell i.e. there was 37.5% more refuse in the poor cell. A significant trend established however is that the biologically dependant variables seemed to be washed out in the first wet period while on the other end of the scale nickel and potassium washed out in almost equal quantities in the first and second wet periods. The remainder of the variables seem to be in-between theses two extremes. Although this is as far as this monitoring program can extend for this dissertation, the Department of Civil Engineering will continue to monitor the cells to determine the shape of the long term cumulative pollutant load curves. Of particular interest are the nickel and potassium curves to see if the pollutants continue to wash out and at what

rate. Results to date indicate that although the poor waste did produce greater loads in two key parameters, the percentage change was not as great as the four parameters in which the rich waste produced higher loads.

8.3.4 Temperature

Waste temperatures were taken twice weekly. A copper constantan thermocouple was placed in the centre of the waste body by driving it in with a metal stake. Once in place the stake was removed, the hole covered and the free end of the thermocouple run to the free end of each cell. Temperatures were taken by means of a thermocouple bridge. See Figure 8.19. Initially it was expected that due to the high organic content in the rich waste the temperature differential would be higher in the initial aerobic phase. The temperature curves for the wastes stayed very close to each other with no discernible temperature peaks due to increased bacteriological action. There was an initial peak in both curves to a temperature near 35°C, this however tapered off and the waste temperature was maintained in the range of 15 to 25 °C for the duration of the experiment. The winter months however, caused a drop in the temperature with the lowest temperature of 5 °C being recorded in the waste body during this period.

8.3.5 Gas

The gas collection devices were constructed and inserted into the covering material overlying the two waste bodies. When gas samples were taken, a gas collection tube (gas bomb) was connected to the collection device and allowed to freely vent for a day. When the samples were taken a pipette pump was used to further pump approximately 5 volumes of the gas bomb through it before it was sealed. When the samples were analysed however, it became evident that the gas samples were being contaminated by atmospheric gas due to the collection procedure. A PhD student is currently working on a device to collect and sample landfill gas off the landfill cover surface.

5.4 CONCLUSIONS

The large scale decomposition data from the test cells, especially in terms of water balance and pollutant loads, proved invaluable in gaining an understanding of the problem. When the tests were started significantly greater differences in the products of decomposition were expected. The differences that did manifest themselves however, will lead to conclusions being drawn. In order to gain further insight into the decomposition processes, the landfill gas produced by the two wastes needs to be established. The lysimeters prove valuable in providing this data.

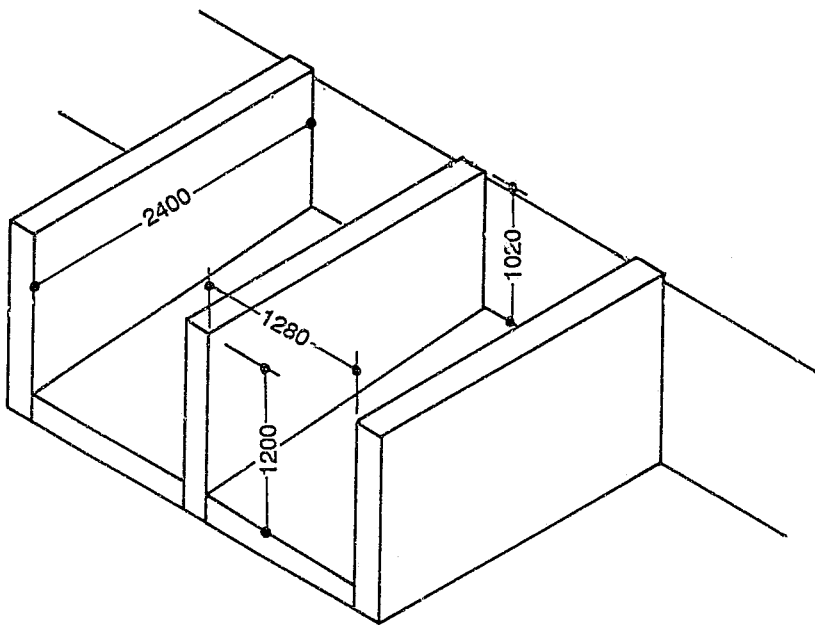


Figure 8.1 Diagram of test cells

CUMULATIVE WATER BALANCE TEST CELLS

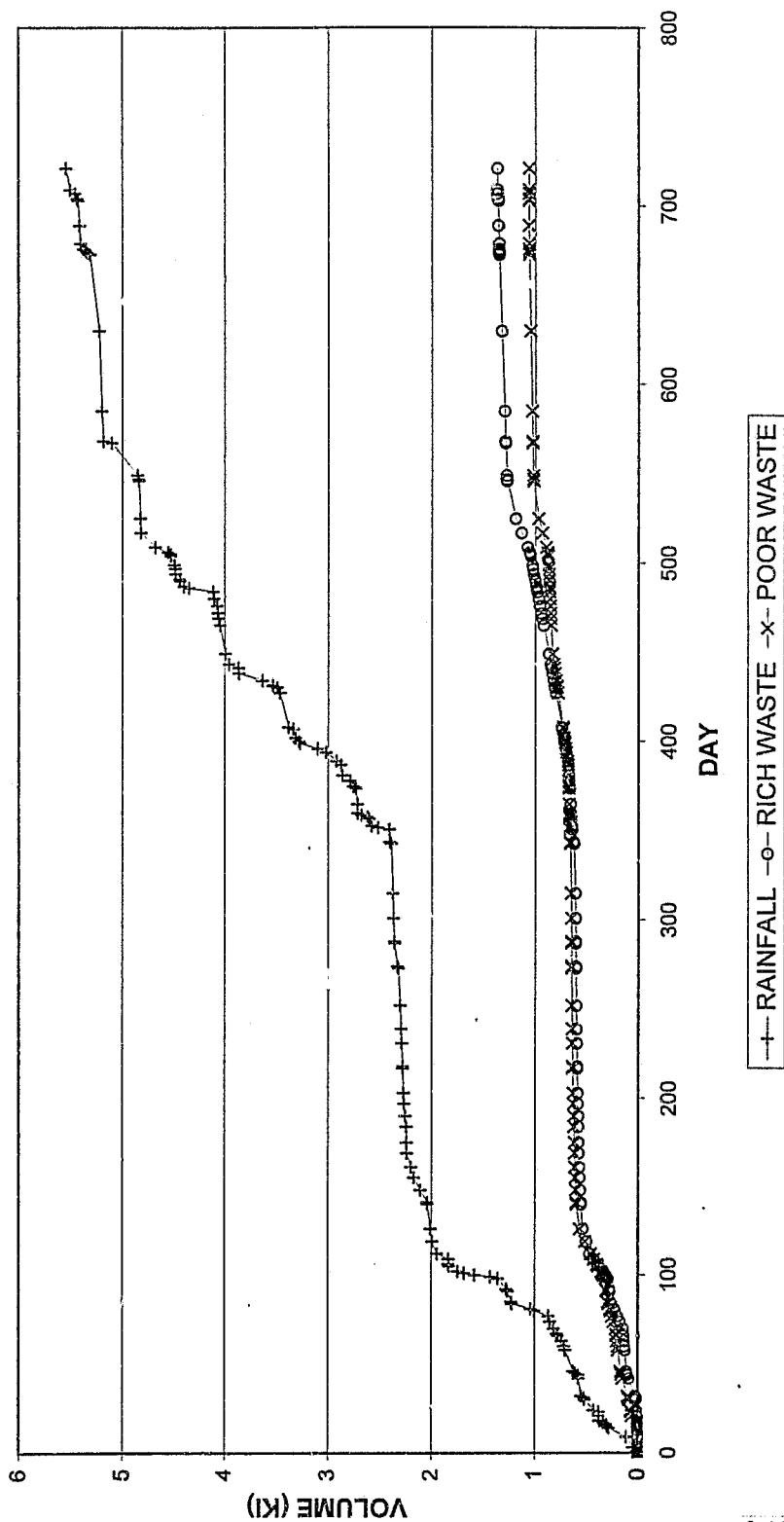


Figure 3.2 Cumulative water balance

LEACHATE PH

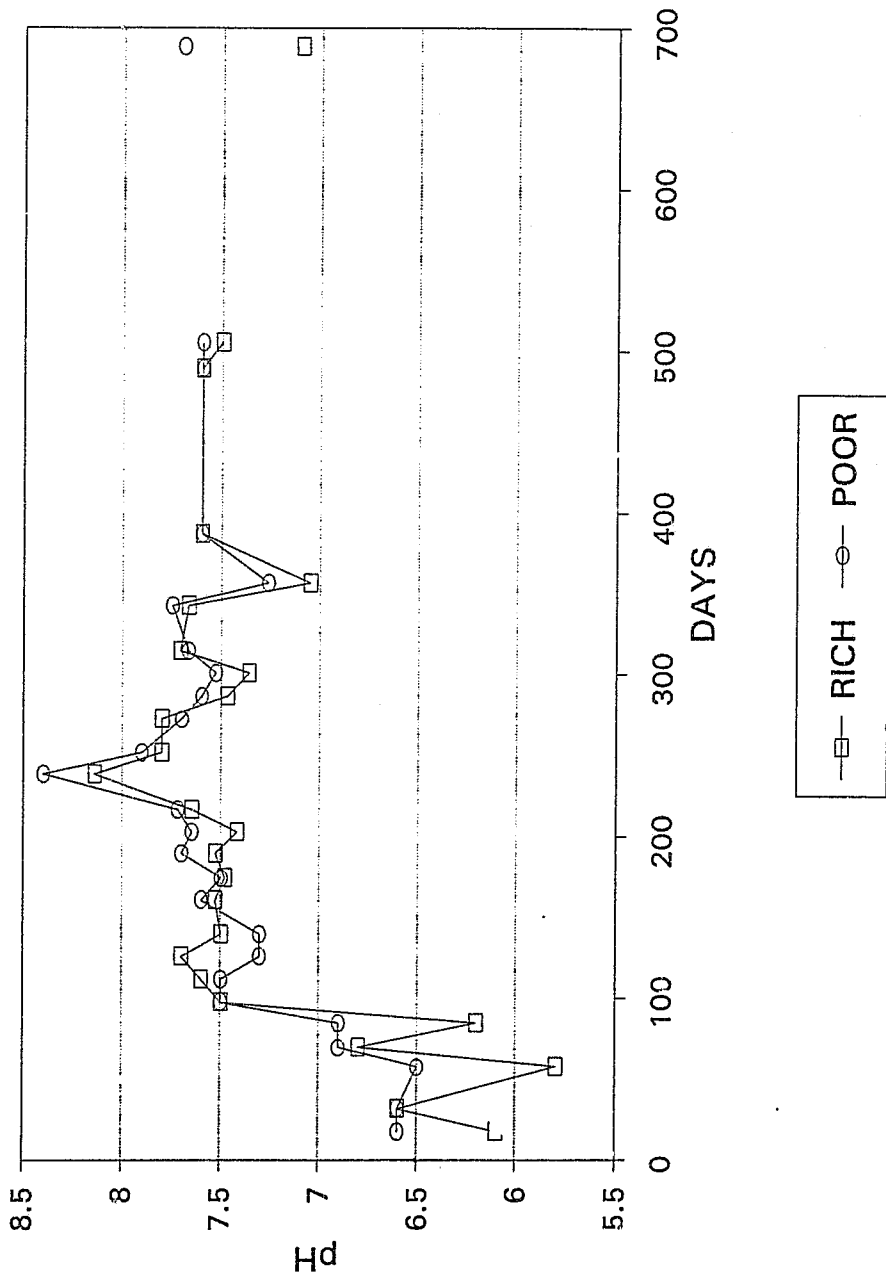


Figure 8.3 Leachate pH

LEACHATE COD

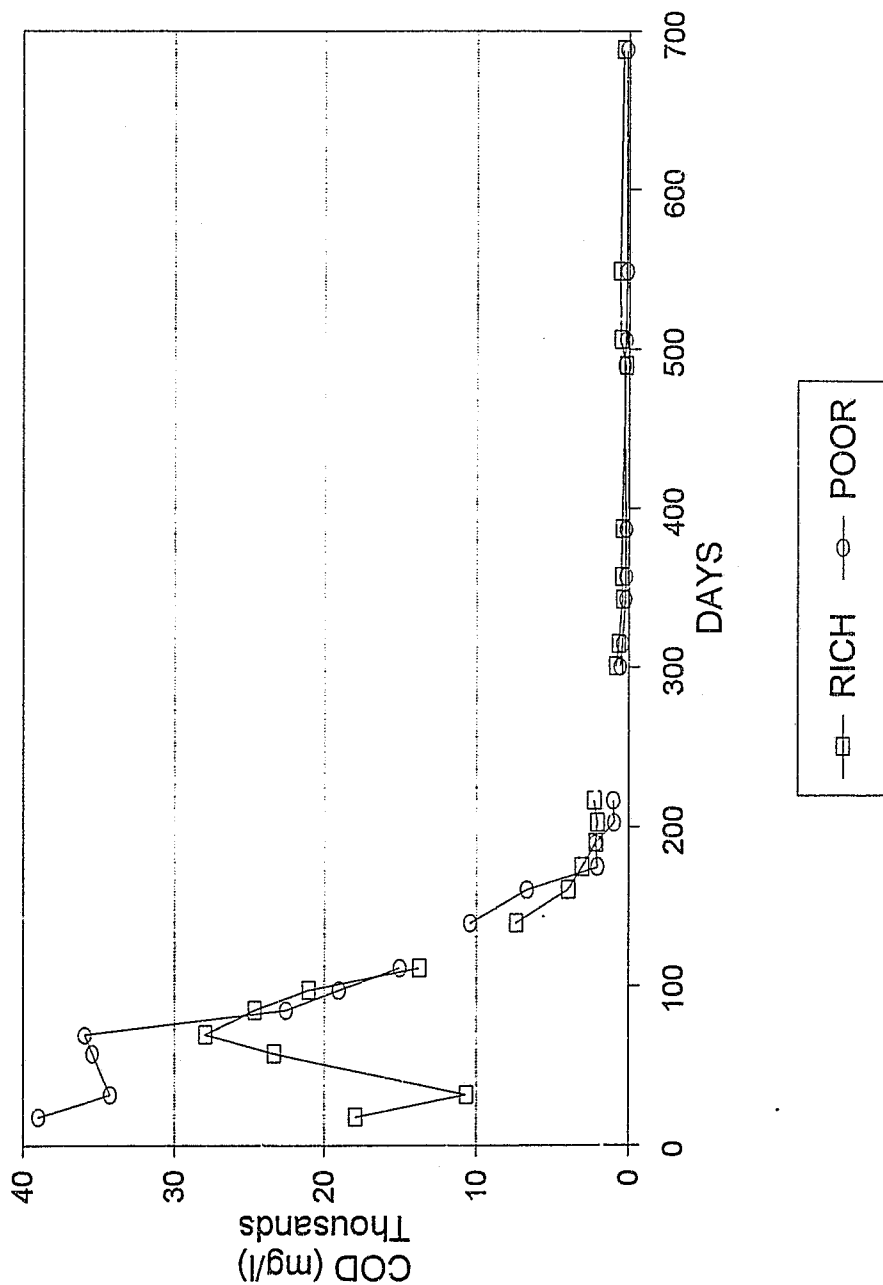


Figure 8.4 Leachate COD

LEACHATE ALKALINITY

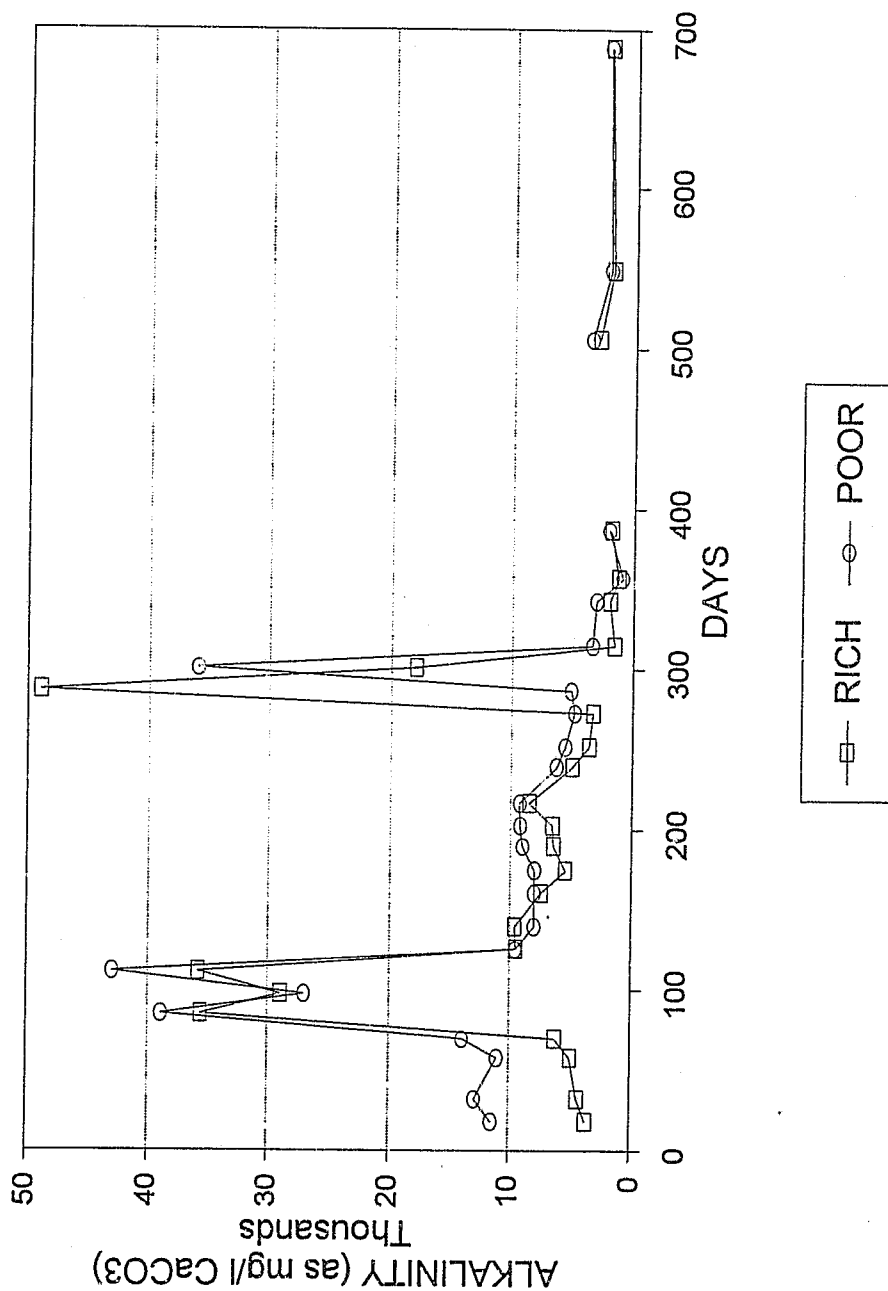


Figure 8.5 Leachate alkalinity

LEACHATE CONDUCTIVITY

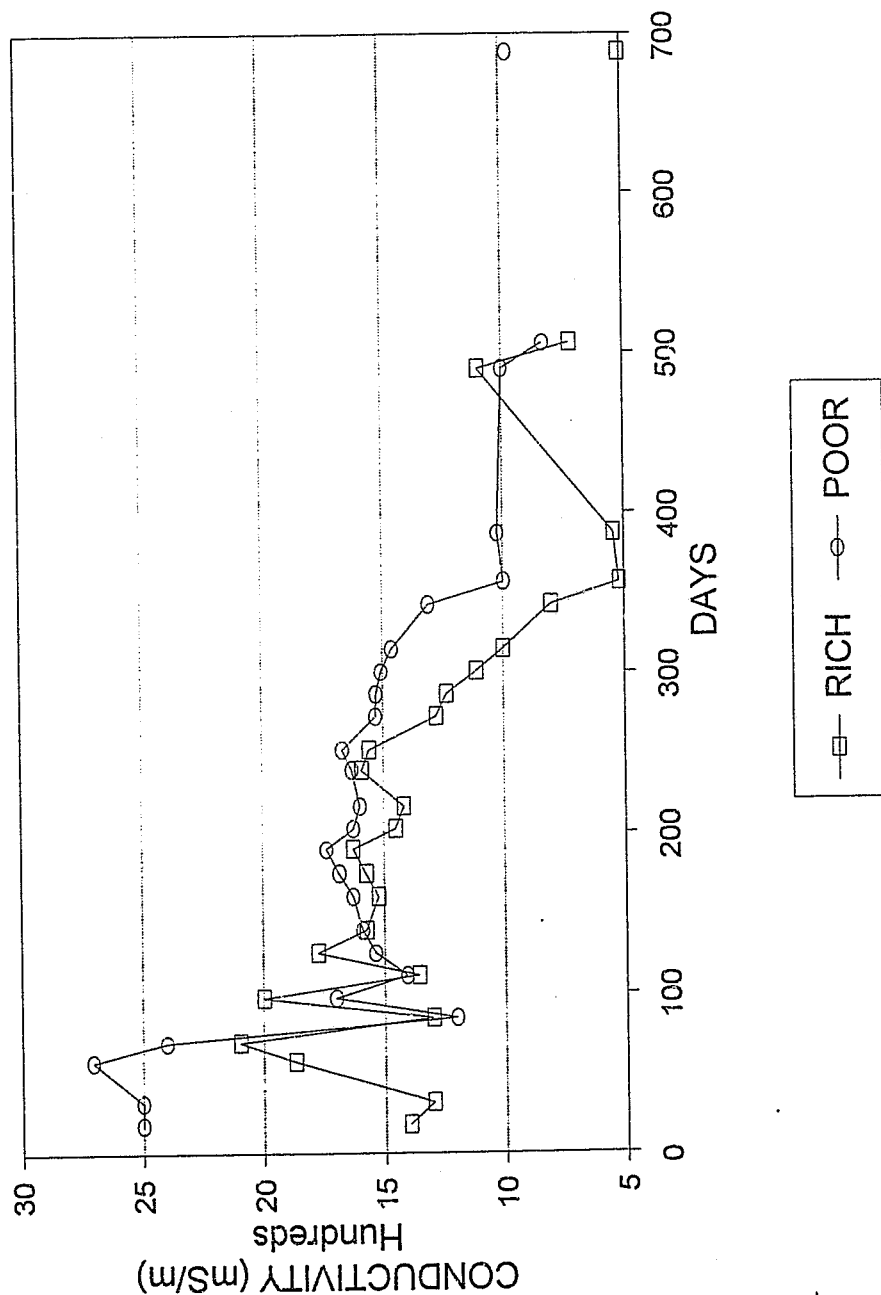


Figure 8.6 Leachate conductivity

LEACHATE CHLORIDE

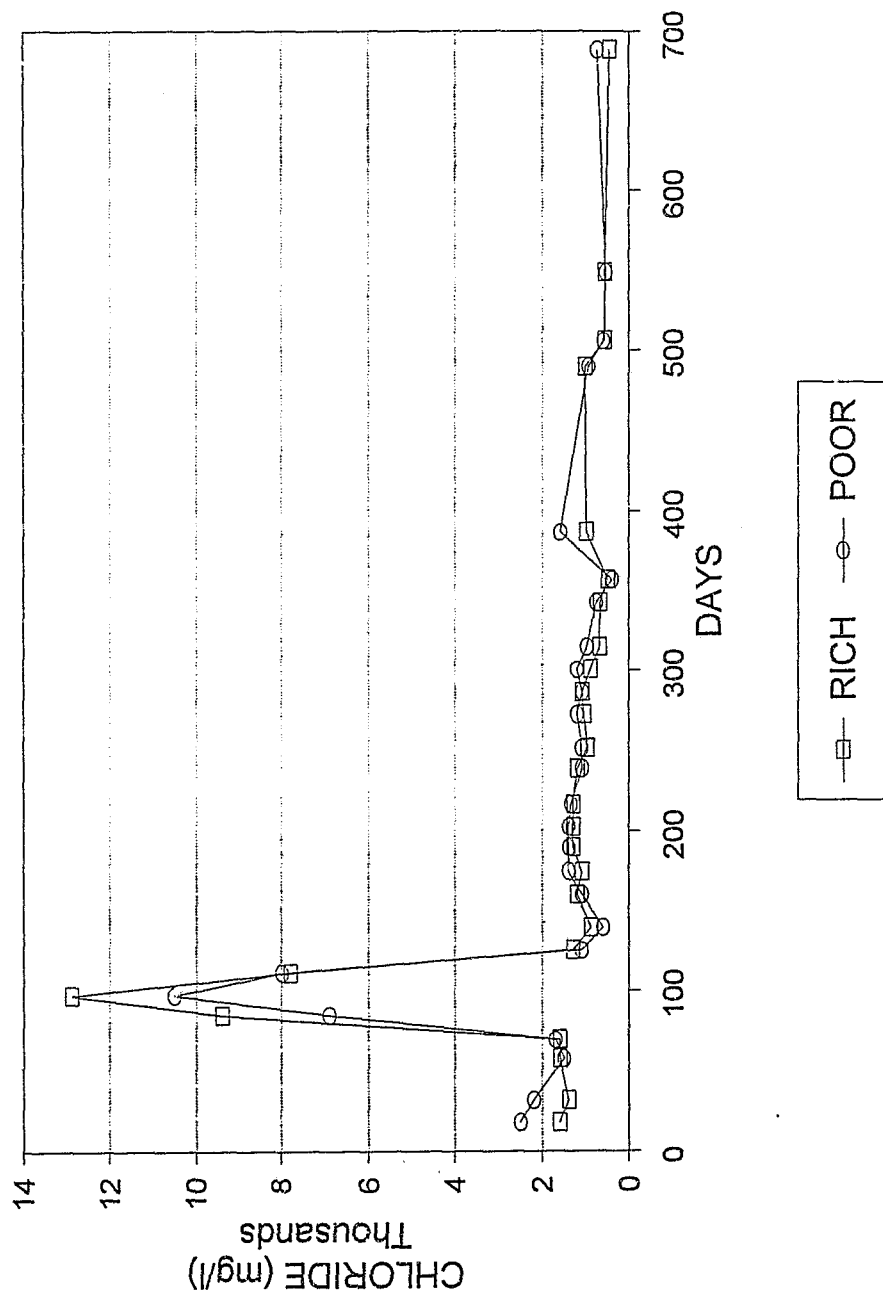


Figure 8.7 Leachate chloride

LEACHATE AMMONIA

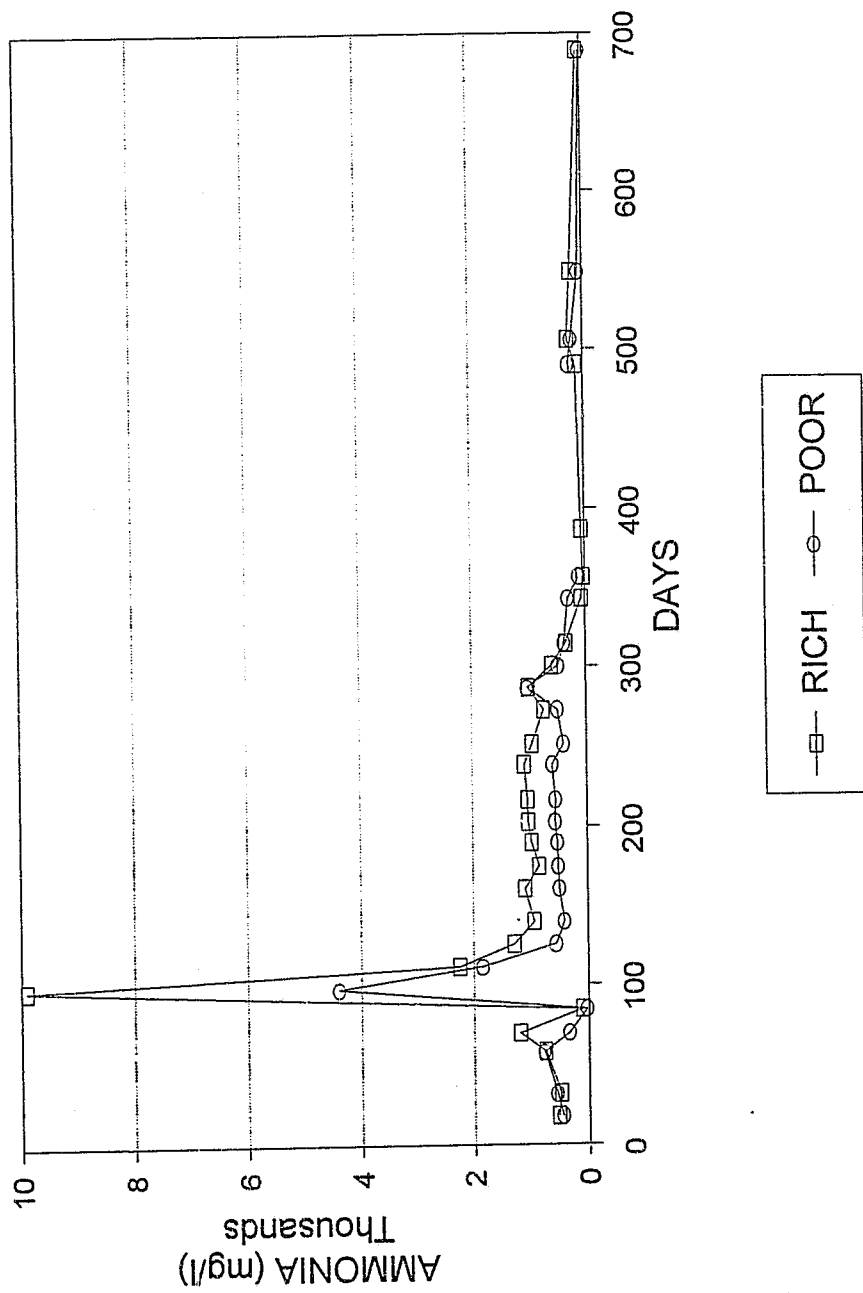


Figure 8.8 Leachate ammonia

LEACHATE NITRATE

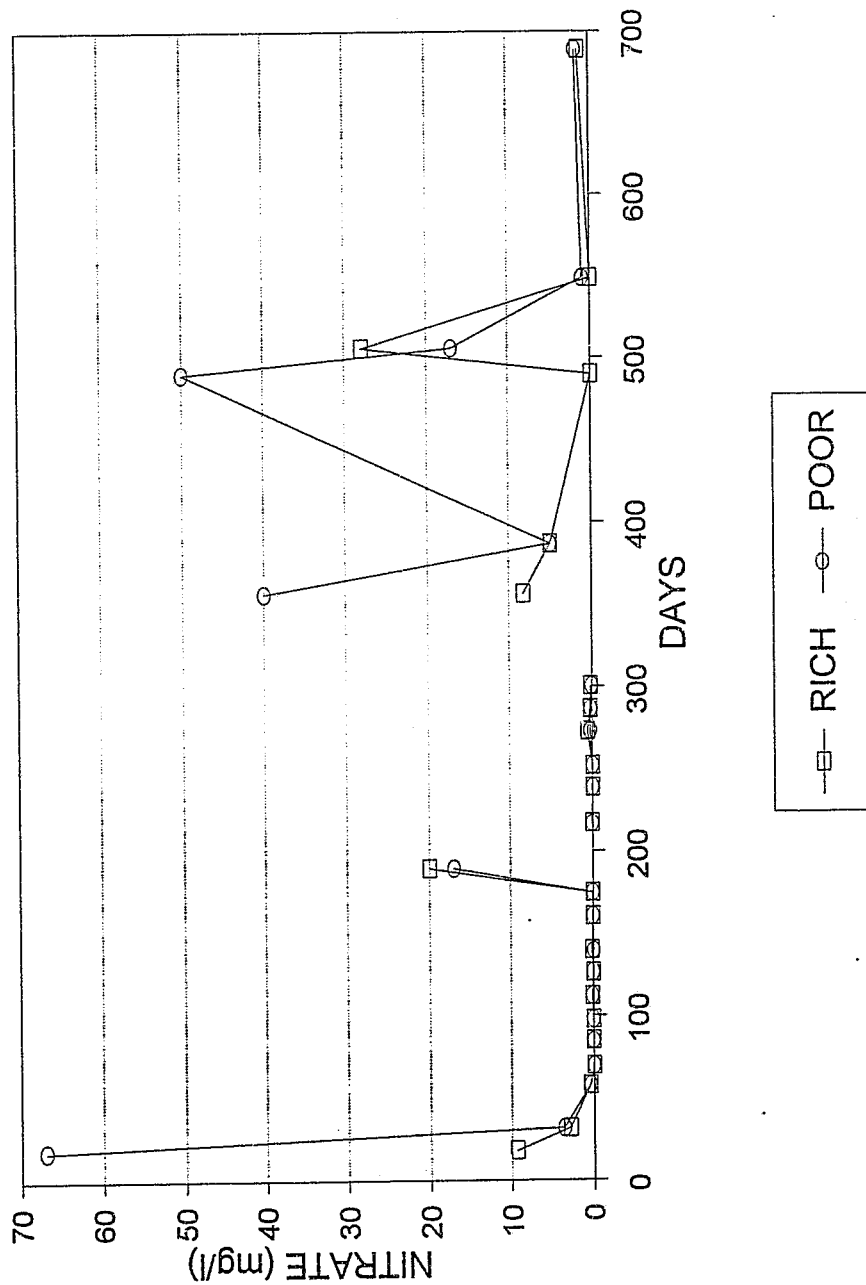


Figure 8.9 Leachate nitrate

LEACHATE NICKEL

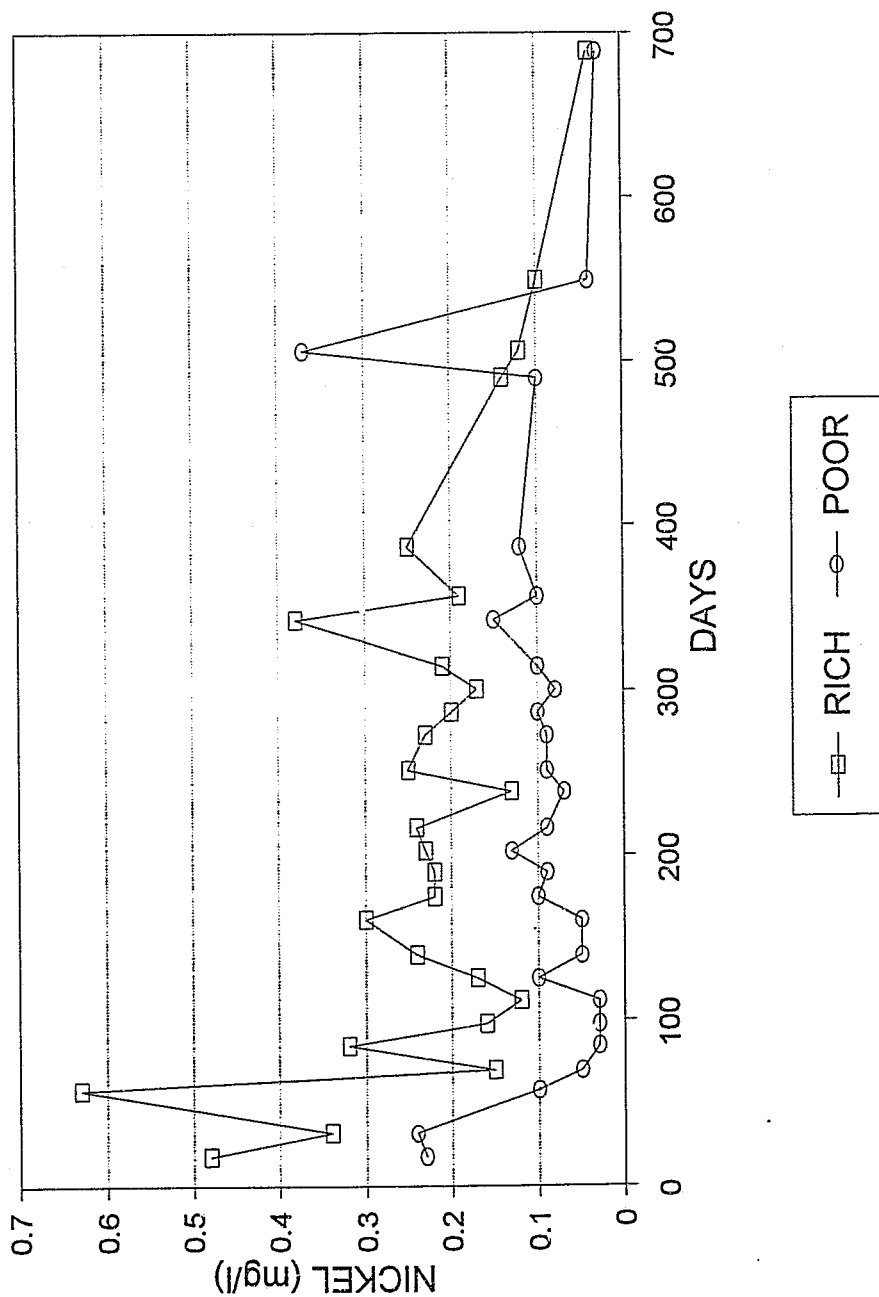


Figure 8.10 Leachate nickel

LEACHATE POTASSIUM

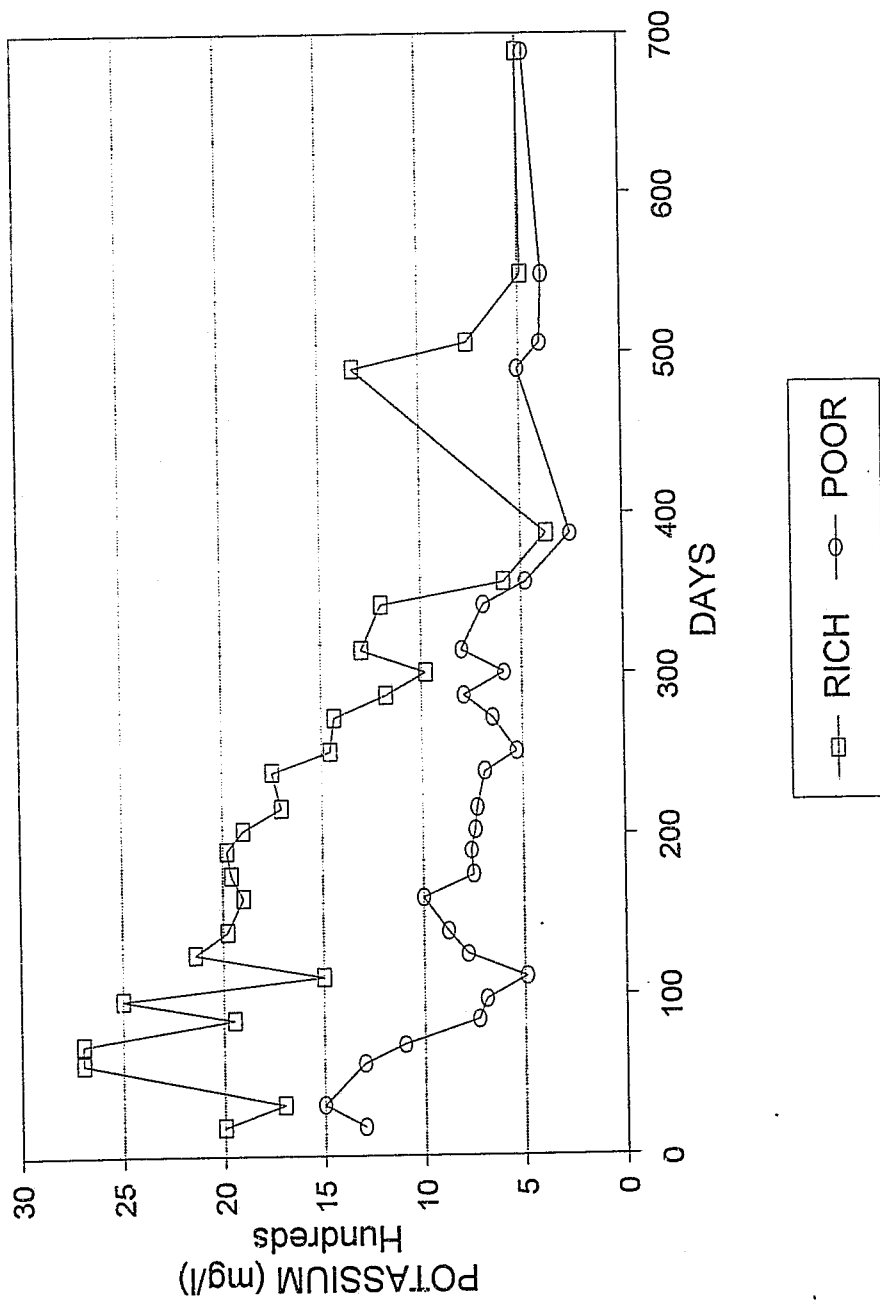


Figure 8.11 Leachate potassium

POLLUTANT LOAD COD

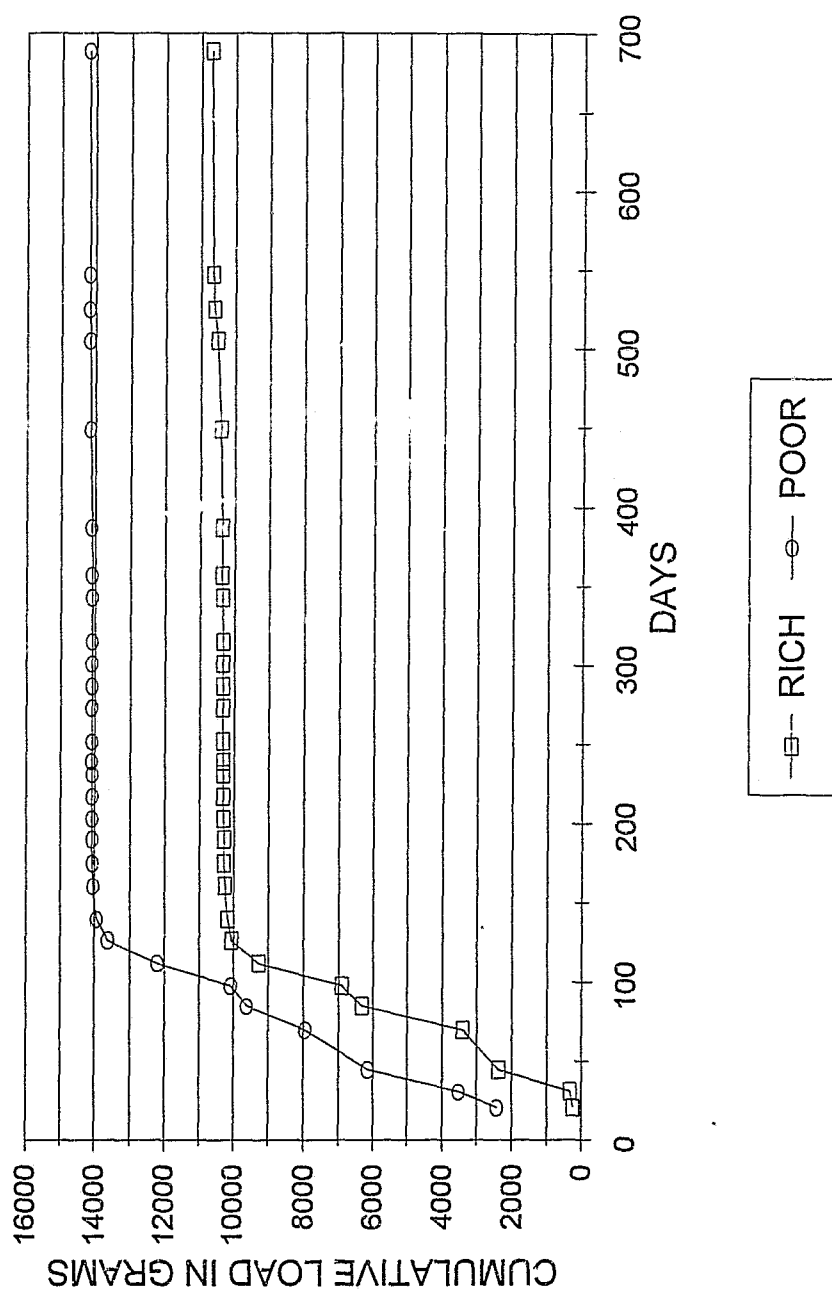


Figure 8.12 Pollutant load COD

POLLUTANT LOAD ALKALINITY

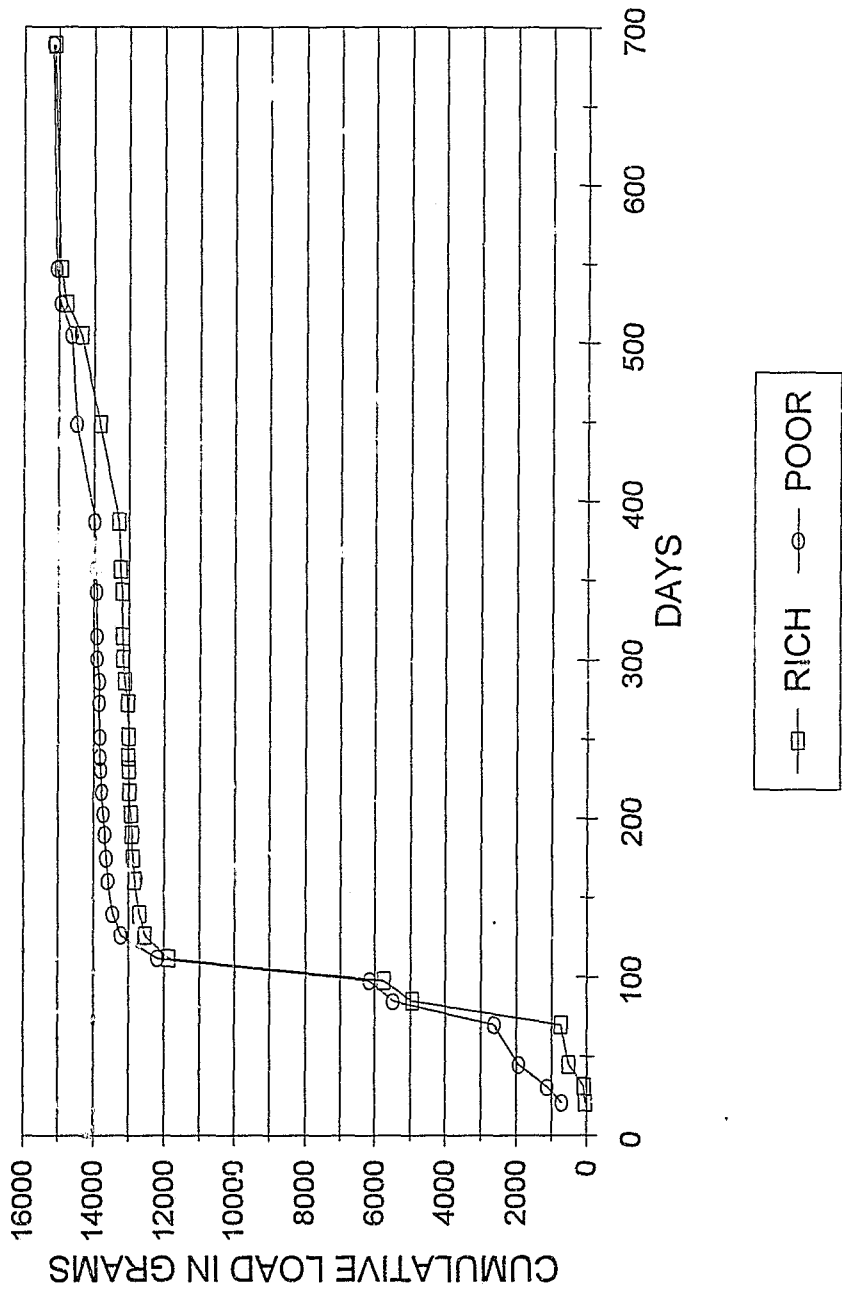


Figure 8.13 Pollutant load alkalinity

POLLUTANT LOAD CHLORIDE

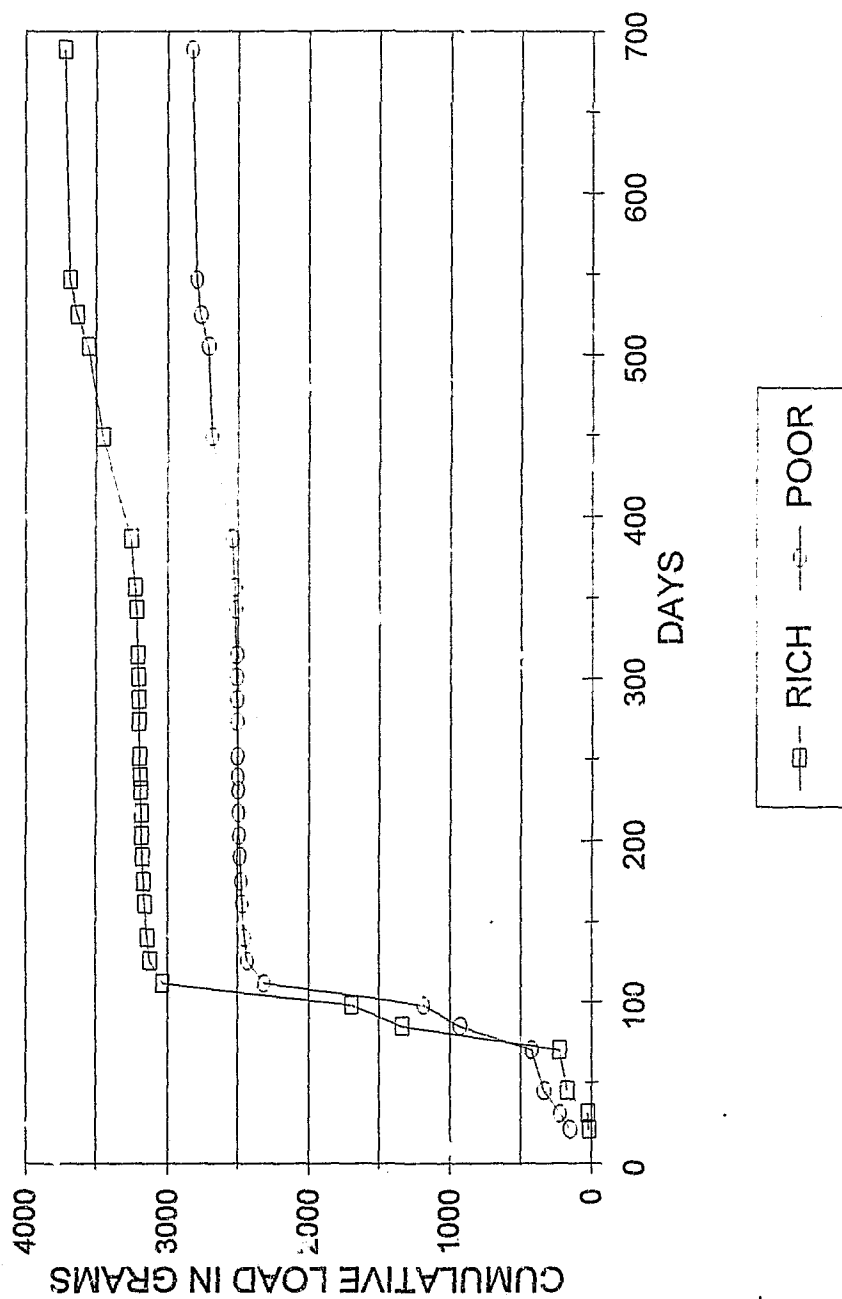


Figure 8.14 Pollutant load chloride

POLLUTANT LOAD

AMMONIA

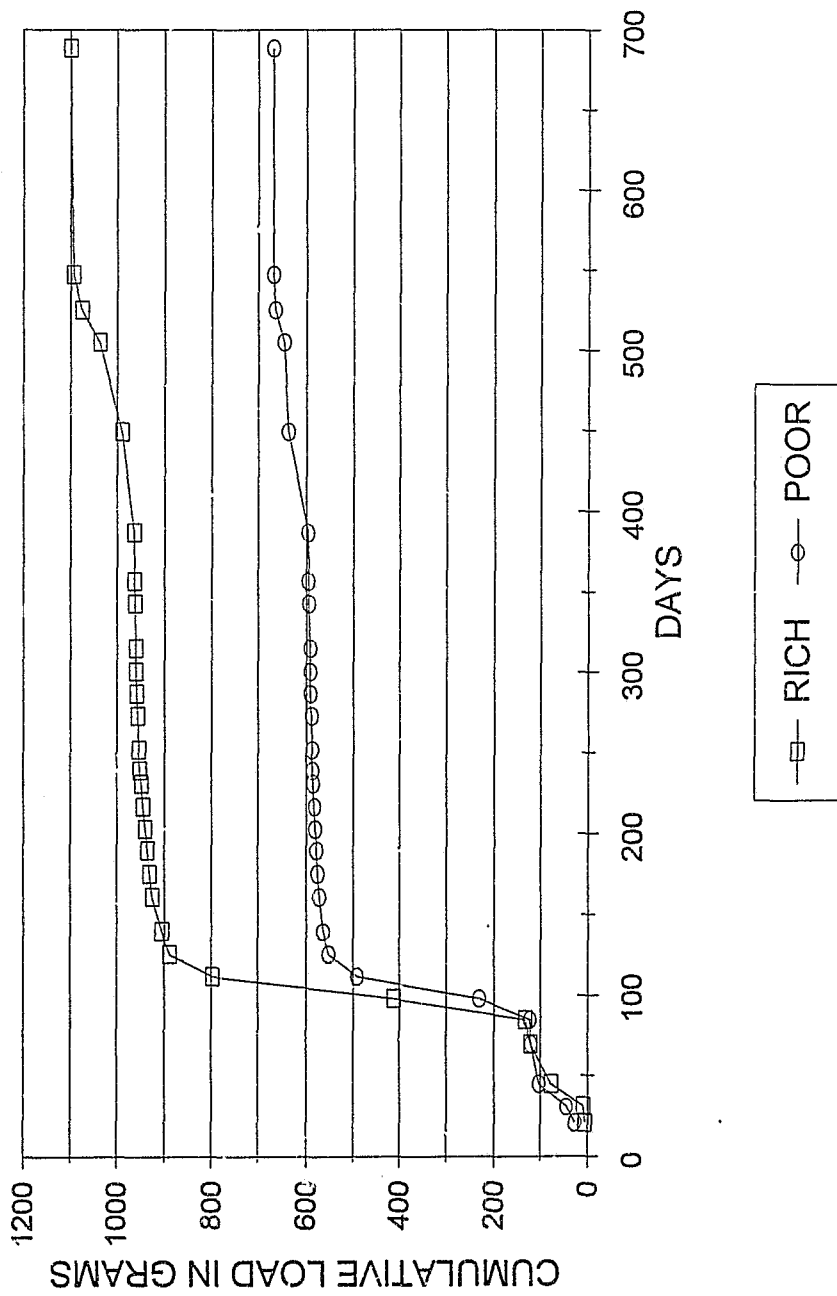


Figure 8.15 Pollutant load ammonia

POLLUTANT LOADS

NITRATE

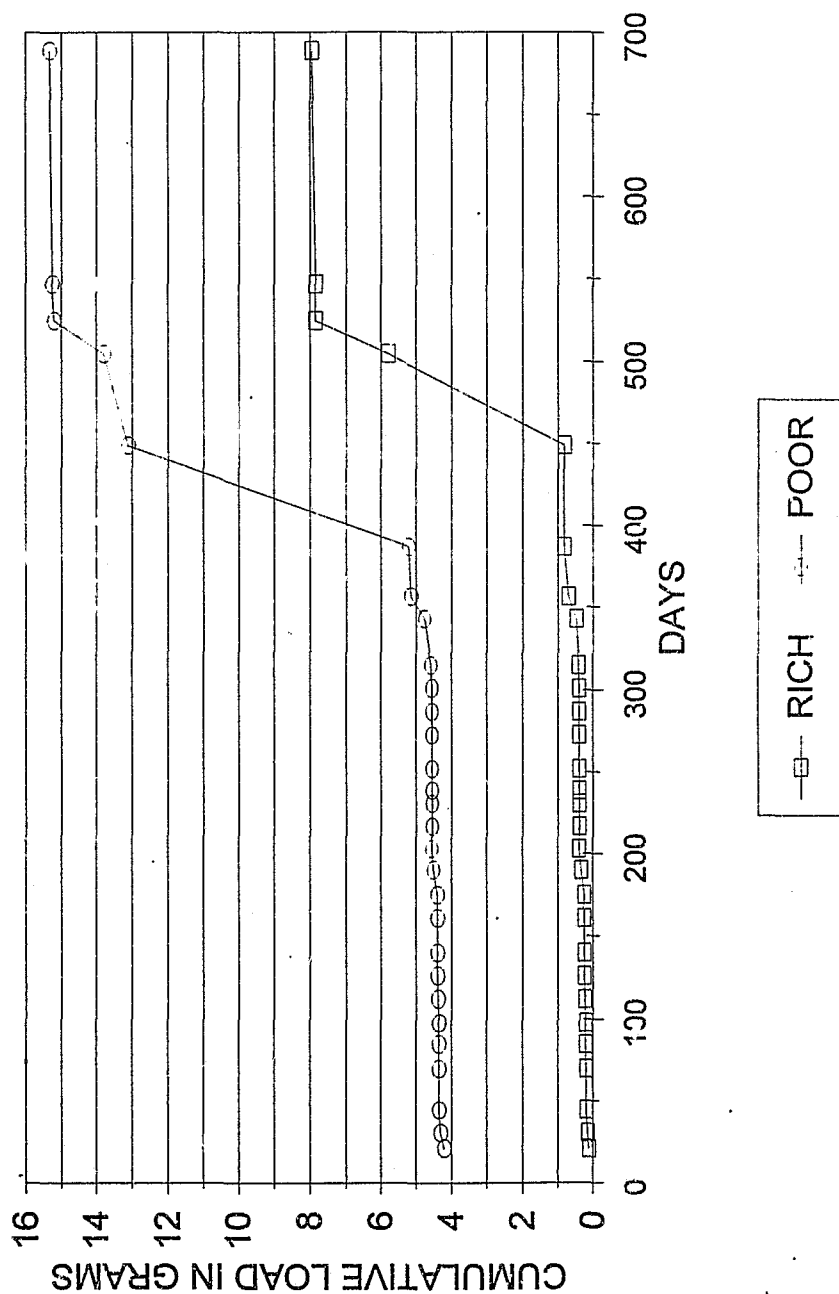


Figure 8.16 Pollutant load nitrate

POLLUTANT LOAD

NICKEL

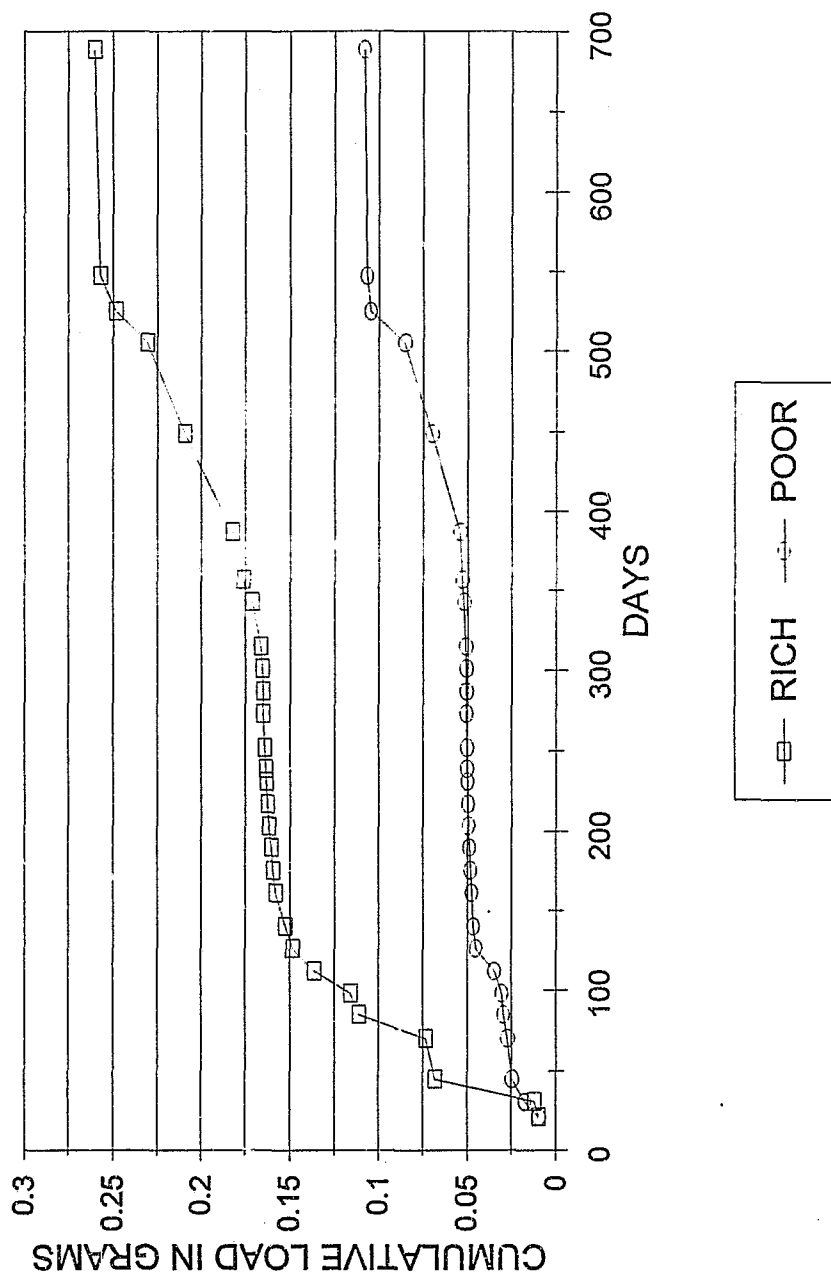


Figure 8.17 Pollutant load nickel

POLLUTANT LOAD POTASSIUM

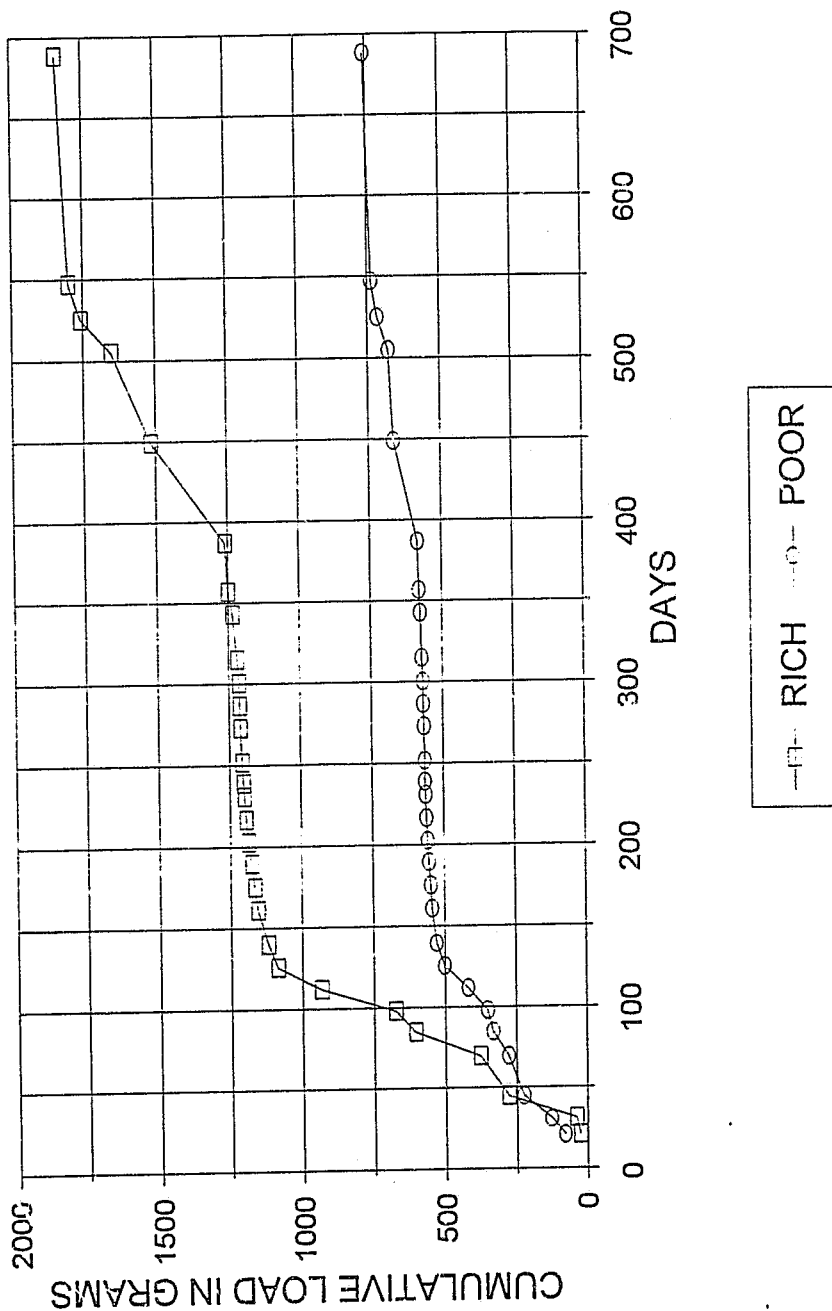


Figure 8.18 Pollutant load potassium

WASTE TEMPERATURES

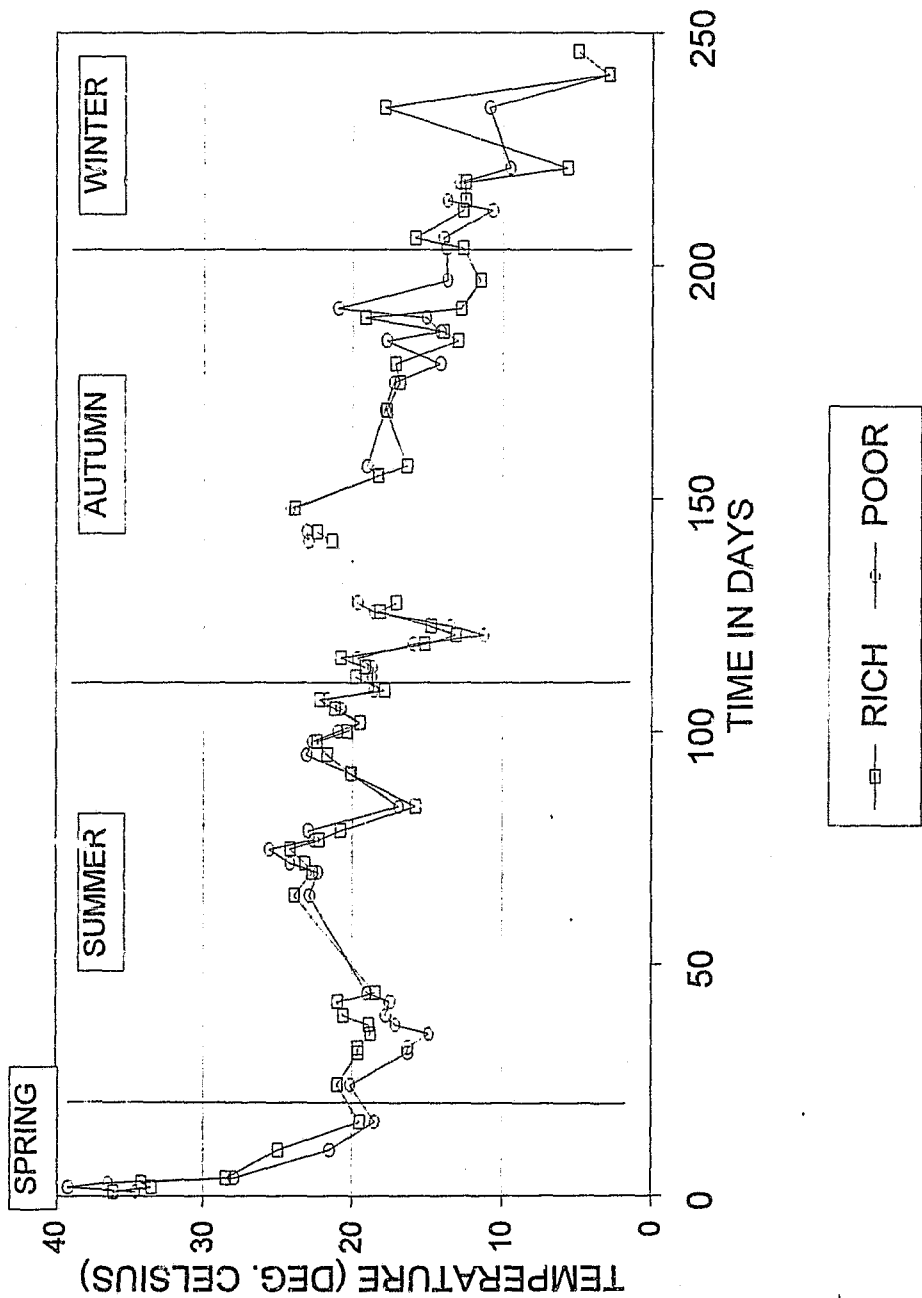


Figure 8.19 Waste temperatures

Table 8.1 Water balance for test cells

Water Balance for Hillman Test Cells.								
Rich Waste Density = 587kg/m ³ , initial MC = 67% dry weight								
Poor Waste Density = 807kg/m ³ , initial MC = 38% dry weight								
Test Cell Area = 3.0725m ²								
Day 0 is 06.11.95								
Date	Day	Rainfall (mm)	Rain Vol (l)	Cum Vol Rain (l)	Leach Rich (l)	Cum Lch Rich (l)	Leach Poor (l)	Cum Lch Poor (l)
06/11/95	0	0	0.00	0	0	0	0	0
07/11/95	1	4	12.29	12.29				
09/11/95	3	12	36.87	49.16				
15/11/95	9	21	64.52	113.68				
20/11/95	14	55	168.99	282.67				
21/11/95	15	5	15.36	298.03				
22/11/95	16	2	6.15	304.18				
23/11/95	17	10	30.73	334.90				
24/11/95	18	12	36.87	371.77	2	2	2	2
27/11/95	21				11.8	13.8	60.7	62.7
29/11/95	23	2	6.15	377.92				
30/11/95	24	16	49.16	427.08				
06/12/95	30	29	89.10	516.18				
07/12/95	31	1.5	4.61	520.79	6.8	20.6	31.4	94.1
08/12/95	32	9	27.65	548.44				
18/12/95	42	8	24.58	573.02				
20/12/95	44	1	3.07	576.09				
21/12/95	45	9	27.65	603.75	88.9	109.5	74	168.1
22/12/95	46	7.5	23.04	626.79				
03/01/96	58	25	76.81	703.60				
05/01/96	60	4	12.29	715.89				
09/01/96	63	7.5	23.04	738.94				
11/01/96	66	12	36.87	775.81				
12/01/96	67	3	9.22	785.02				
15/01/96	70				36.1	145.6	50	218.1
19/01/96	74	22	67.60	852.62				
22/01/96	77	5	15.36	867.98				
25/01/96	80	44	135.19	1003.17				
26/01/96	81	12	36.87	1040.04				
29/01/96	84	60	184.35	1224.39				
30/01/96	85				117.9	263.5	73.7	291.8
05/02/96	91	13	39.94	1264.33				
06/02/96	92	3.5	10.75	1275.09				
12/02/96	98	27	82.96	1358.05	28.1	291.6	24.7	316.5
13/02/96	99	24	73.74	1431.79				
14/02/96	100	49	150.55	1582.34				
15/02/96	101	33	101.33	1683.73				
16/02/96	102	20	61.45	1745.18				
19/02/96	105	27	82.96	1828.14				
20/02/96	106	2	6.15	1834.28				
23/02/96	109	0.5	1.54	1835.82				
26/02/96	112	35	107.54	1943.36	170.9	462.5	140.4	456.9
04/03/96	119	16	49.16	1992.52				
11/03/96	126				70.7	533.2	107.3	564.2
25/03/96	140				17.7	550.9	31.1	595.3
26/03/96	141	15	46.09	2038.60				
02/04/96	148	22	67.60	2106.20				
09/04/96	155	20	61.45	2167.65				
15/04/96	161				17.8	568.7	16.1	611.4

23/04/96	169	22	67.60	2235.24				
29/04/96	175				6.4	575.1	5.1	616.5
08/05/96	184	2	6.15	2241.39				
14/05/96	190				4.4	579.5	5.4	621.9
21/05/96	197	7.5	23.04	2264.43				
27/05/96	203				5.4	584.9	5.3	627.2
10/06/96	217				4.3	589.2	4.9	632.1
11/06/96	218	3.5	10.75	2275.19				
24/06/96	231				3.5	592.7	2.7	634.8
02/07/96	239				2.7	595.4	3.2	638
15/07/96	252				2	597.4	1.9	639.9
05/08/96	273	14	43.02	2318.20	3.4	600.8	2.3	642.2
06/08/96	274	3	9.22	2327.42				
19/08/96	287				1.8	602.6	1.5	643.7
20/08/96	288	10	30.73	2358.14				
02/09/96	301				2.5	605.1	1.8	645.5
16/09/96	315				3.4	608.5	2.5	648
14/10/96	343	10	30.73	2388.87	12.6	621.1	6.4	654.4
15/10/96	344	5	15.36	2404.23				
22/10/96	351	1	3.07	2407.30				
23/10/96	352	35	107.54	2514.84				
24/10/96	353	20	61.45	2576.29				
28/10/96	357	8	24.58	2600.87	26.1	647.2	9.8	664.2
29/10/96	358	5	15.36	2616.23				
30/10/96	359	20	61.45	2677.68				
31/10/96	360	11	33.80	2711.48				
05/11/96	365	1	3.07	2714.55				
14/11/96	374	5	15.36	2729.92				
15/11/96	375	8	24.58	2754.50				
18/11/96	378	10	30.73	2785.22				
21/11/96	381	24	73.74	2858.96				
27/11/96	387	5	15.36	2874.32	25	672.2	11.2	675.4
29/11/96	389	14	43.02	2917.34				
04/12/96	394	33	101.39	3018.73				
06/12/96	396	26	79.89	3098.62				
09/12/96	399	56	172.06	3270.68				
10/12/96	400	3	9.22	3279.89				
12/12/96	402	10	30.73	3310.62				
17/12/96	407	8	24.58	3335.20				
18/12/96	408	15	46.09	3381.29				
06/01/97	427	26	79.89	3461.17				
09/01/97	430	9	27.65	3488.82				
10/01/97	431	13	39.94	3528.77				
13/01/97	434	33	101.39	3630.16				
17/01/97	438	74	227.37	3857.52				
20/01/97	441	2	6.15	3863.67				
22/01/97	443	30	92.18	3955.84				
28/01/97	449	12	36.87	3992.71	193.9	866.1	158.1	833.5
13/02/97	465	17	52.23	4044.95				
17/02/97	469	5	15.36	4060.31				
20/02/97	472	3.5	10.75	4071.06				
24/02/97	476	1.5	4.61	4075.67				
28/02/97	480	10	30.73	4108.40				
04/03/97	484	3	9.22	4115.61				
06/03/97	486	78	239.66	4355.27				
07/03/97	487	17	52.23	4407.50				
10/03/97	490	11	33.80	4441.30				
11/03/97	491	3	9.22	4450.52				
14/03/97	494	12	36.87	4487.39				

17/03/97	497	1	3.07	4490.46				
19/03/97	499	3	9.22	4499.68				
24/03/97	504	11	33.80	4533.47				
25/03/97	505	0.5	1.54	4535.01	177.6	1043.7	40.6	874.1
26/03/97	506	8	24.58	4559.59				
29/03/97	509	40	122.90	4682.49				
06/04/97	517	45	138.26	4820.75				
14/04/97	525				146	1189.7	93.8	967.9
05/05/97	546	6	18.44	4839.19				
06/05/97	547				85.9	1275.6	49.3	1017.2
08/05/97	549	5	15.36	4854.55				
26/05/97	567	80	245.80	5100.35				
27/05/97	568	26	79.89	5180.24				
13/06/97	585	5	15.36	5195.60				
28/07/97	630	8	24.58	5220.18				
09/09/97	673	28	86.03	5306.21				
10/09/97	674	8	24.58	5330.79				
11/09/97	675	6	18.44	5349.22				
12/09/97	676	7	21.51	5370.73				
15/09/97	679	9	27.65	5398.38				
25/09/97	689				81.7	1357.3	45.5	1062.7
09/10/97	703	8.5	26.12	5424.50				
10/10/97	704	3.5	10.75	5435.25				
13/10/97	707	6.5	19.97	5455.22	11.8	1369.1	0	1062.7
15/10/97	709	15	46.09	5501.31				
27/10/97	721	13	39.94	5541.25				

Table 8.2 Leachate analysis for poor waste

TEST CELL LEACHATE POOR WASTE									
DAY	PH	COD	ALK	CONDUCT	CL	AMM	NITRATE	NI	POT
18	6.6	39000	11400	2500	2500	460	67	0.23	1300
32	6.6	34300	12800	2500	2200	560	3.6	0.24	1500
58	6.5	35500	11000	2710	1500	780	0.5	0.1	1300
70	6.9	36000	13900	2400	1700	340	0.03	0.05	1100
85	6.9	22600	38900	1200	6900	34	0.1	0.03	730
98	7.5	19100	27000	1700	10500	4400	0.1	0.03	690
112	7.5	15100	43000	1408	8000	1850	0.2	0.03	490
126	7.3		9500	1538	1100	560	0.12	0.1	780
140	7.3	10400	8000	1589	610	410	0.02	0.05	880
161	7.6	6700	8000	1629	1100	490	0.1	0.05	1000
175	7.5	2080	8000	1685	1400	510	0.1	0.1	750
190	7.7	2200	9000	1737	1400	520	17	0.09	760
203	7.65	990	9200	1625	1400	560		0.13	740
217	7.72	1040	9250	1598	1350	550	0.1	0.09	728
239	8.4		6300	1630	1100	600	0.05	0.07	690
252	7.9		5600	1670	1160	400	0.02	0.09	530
273	7.7		4800	1530	1200	500	0.3	0.09	650
287	7.6		5100	1528	1100	1040	0.2	0.1	790
301	7.53	610	36000	1507	1200	480	0.1	0.08	590
315	7.67	440	3400	1464	990	370		0.1	800
343	7.75	260	3100	1311	770	300		0.15	690
357	7.26	220	980	997	400	110	40	0.1	480
387	7.6	240	2100	1021	1600		5	0.12	260
400	7.6	340		1000	950	260	50	0.1	510
506	7.6	210	3600	830	600	210	17	0.37	400
549		150	2120		550	76	1	0.04	390
689	7.7	160	2280	970	740	20	1.7	0.03	470

Table 8.3 Leachate analysis for rich waste

TEST CELL LEACHATE
RICH WASTE

DAY	PH	COD	ALK	CONDUC	CL	ANMM	NITRATE	NI	POT
18	6.1	18000	3700	1400	1600	530	9.4	0.48	2000
32	6.3	10700	4400	1300	1400	490	2.9	0.34	1700
58	5.8	23400	5000	1870	1600	760	0.5	0.63	2700
70	6.8	28000	6300	2100	1600	1200	0.03	0.15	2700
85	6.2	24700	35600	1300	9400	110	0.1	0.32	1950
98	7.5	21100	29000	2000	12900	9900	0.1	0.16	2500
112	7.6	13800	35800	1358	7800	2250	0.2	0.12	1500
126	7.7		9500	1773	1300	1300	0.12	0.17	2140
140	7.5	7400	9500	1575	890	950	0.2	0.24	1980
161	7.53	4000	7500	1525	1200	1100	0.1	0.3	1900
175	7.48	3100	5500	1574	1100	850	0.1	0.22	1960
190	7.53	2200	6500	1627	1300	980	20	0.22	1980
203	7.42	2100	6600	1453	1300	1040		0.23	1900
217	7.65	2300	8500	1418	1300	1050	0.1	0.24	1706
239	8.14		5000	1590	1200	1100	0.05	0.13	1750
252	7.8		3600	1560	970	960	0.02	0.25	1460
273	7.8		3300	1280	1050	750	0.6	0.23	1440
287	7.47		49000	1237	1100	1020	0.3	0.2	1180
301	7.36	890	18300	1112	900	600	0.2	0.17	980
315	7.71	720	1500	1003	700	350		0.21	1300
343	7.67	440	2000	803	680	69		0.38	1200
357	7.05	500	1300	522	500	35	8.3	0.19	590
387	7.6	480	1900	546	1000	64	5	0.25	380
490	7.6	220		1100	1020	130	0	0.14	1330
506	7.5	580	3000	720	580	270	28	0.12	760
549		640	1930		580	220	0	0.1	490
589	7.1	400	2160	510	460	67	1.4	0.04	500

Table 8.4 Pollutant loads for poor waste

TEST CELL POLLUTANT LOADS
POOR WASTE

DAY	LEACH /	CUM LCH /	POOR WASTE LEACHATE QUALITY (mg/l)						POOR WASTE LEACHATE LOADS			
			COD	ALK	CL	AMM	NITRAT ¹	jl	POT	COD	ALK	CL
21	62.7	62.7	39000	11400	2500	460	67	0.23	1300	2445.30	714.78	156.75
31	31.4	94.1	34300	12800	2200	560	3.6	0.24	1500	1077.02	401.92	69.08
45	74.0	168.1	35500	11000	1500	780	0.5	0.1	1300	2627.00	814.00	111.00
70	50.0	218.1	36000	13900	1700	340	0.03	0.05	1100	1800.00	695.00	85.00
85	73.7	291.8	22600	38900	6900	34	0.1	0.03	730	1666.07	2867.71	508.67
98	24.7	316.5	19100	27000	10500	4400	0.1	0.03	690	471.01	665.82	258.93
112	140.4	456.8	15100	43000	8000	1850	0.2	0.03	490	2119.44	6035.48	1122.88
126	137.3	564.2	13220	9500	1100	560	0.12	0.1	780	1418.64	1019.45	118.04
140	31.1	595.2	10400	8000	610	410	0.02	0.05	880	323.23	248.64	18.96
161	16.1	611.4	6700	8000	1100	490	0.1	0.05	1000	108.00	128.96	17.73
175	5.1	616.4	2080	8000	1400	510	0.1	0.1	750	10.55	40.56	7.10
190	5.4	621.8	2200	9000	1400	520	17	0.09	760	11.88	48.60	7.56
203	5.3	627.1	990	9200	1400	580	9.2	0.13	740	5.27	48.94	7.45
217	4.9	632.0	1040	9250	1350	550	0.1	0.09	728	5.06	45.05	6.57
231	2.7	634.7	968.3	7283.3	1183.3	583.3	0.07	0.08	709	2.61	19.66	3.19
239	3.2	637.9	927.4	6300	1100	600	0.05	0.07	690	3.00	20.35	3.55
252	1.9	639.9	860.8	5600	1100	400	0.02	0.09	530	1.66	10.81	2.12
273	2.3	642.2	753.3	4800	1200	500	0.3	0.09	650	1.73	11.04	2.76
287	1.5	643.7	681.7	5100	1100	1040	0.2	0.1	790	1.02	7.65	1.65
301	1.8	645.4	610	36000	1200	480	0.1	0.08	590	1.07	63.36	2.11
315	2.5	647.9	440	3400	990	370	10.1	0.1	800	1.08	8.36	2.44
343	6.4	654.3	260	3100	770	300	30	0.15	690	1.66	19.78	4.91
357	9.8	664.1	220	980	400	110	40	0.1	480	2.16	9.64	3.94
387	11.2	675.3	240	2100	1600	148.1	5	0.12	260	2.69	23.56	17.95
449	158.1	833.4	340	3323.7	950	260	50	0.1	510	53.75	525.48	150.20
505	40.6	874.0	210	3600	600	210	17	0.37	400	8.53	146.16	24.36
525	93.8	967.8	202.7	3419.5	553.9	193.7	15	0.21	395	19.01	320.75	55.71
547	49.3	1017.1	150	2120	550	76	1	0.04	390	7.40	104.52	27.12
689	45.5	1062.6	160	2280	740	20	1.7	0.03	470	7.28	103.74	33.67

POOR WASTE LEACHATE CUMULATIVE LOADS (g)											
(g)											
AMM	NITRATE	NI	POT	COD	ALK	CL	AMM	NITRATE	NI	POT	
28.84	4.20	0.01	81.51	2445.30	714.78	156.75	28.84	4.20	0.01	81.51	
17.58	0.11	0.01	47.10	3522.32	1116.70	225.83	46.42	4.31	0.02	128.61	
57.72	0.04	0.01	96.20	6149.32	1930.70	336.83	104.14	4.35	0.02	224.81	
17.00	0.00	0.00	55.00	7949.32	2625.70	421.83	121.14	4.35	0.03	279.81	
2.51	0.01	0.00	53.82	9615.39	5493.41	930.50	123.05	4.36	0.03	333.63	
108.50	0.00	0.00	17.02	10086.40	6159.23	1189.43	232.15	4.36	0.03	350.64	
259.67	0.03	0.00	68.78	12205.83	12194.71	2312.31	491.82	4.39	0.03	419.42	
60.09	0.01	0.01	83.70	13624.47	13214.15	2430.35	551.91	4.40	0.05	503.12	
12.74	0.00	0.00	27.35	13947.70	13482.79	2449.31	564.66	4.40	0.05	530.47	
7.90	0.00	0.00	16.12	14055.71	13591.75	2467.04	572.56	4.40	0.05	546.59	
2.59	0.00	0.00	3.80	14066.25	13632.31	2474.14	575.14	4.41	0.05	550.39	
2.81	0.09	0.00	10	14078.13	13680.91	2481.70	577.95	4.50	0.05	554.50	
2.98	0.05	0.00	3.94	14083.40	13729.86	2489.15	580.93	4.55	0.05	558.43	
2.68	0.00	0.00	3.55	14088.47	13774.90	2495.72	583.61	4.55	0.05	561.98	
1.57	0.00	0.00	1.91	14091.08	13794.57	2498.92	585.18	4.55	0.05	563.89	
1.94	0.00	0.00	2.23	14094.08	13814.92	2502.47	587.12	4.55	0.05	566.12	
0.77	0.00	0.00	1.02	14095.74	13825.73	2504.59	587.89	4.55	0.05	567.14	
1.15	0.00	0.00	1.49	14097.47	13836.77	2507.35	589.04	4.55	0.05	568.64	
1.56	0.00	0.00	1.19	14098.49	13844.42	2509.00	590.60	4.55	0.05	569.82	
0.84	0.00	0.00	1.04	14099.57	13907.78	2511.11	591.45	4.55	0.05	570.86	
0.91	0.02	0.00	1.97	14100.65	13916.14	2513.55	592.36	4.57	0.05	572.83	
1.91	0.19	0.00	4.40	14102.31	13935.92	2518.46	594.27	4.76	0.05	577.23	
1.08	0.39	0.00	4.72	14104.47	13945.56	2522.40	595.35	5.16	0.05	581.96	
1.66	0.06	0.00	2.92	14107.16	13969.12	2540.35	597.02	5.21	0.05	584.57	
41.11	7.91	0.02	80.63	14110.92	14494.60	2690.54	638.12	13.12	0.07	665.50	
8.53	0.69	0.02	16.24	14169.44	14640.76	2714.90	646.65	13.81	0.09	681.74	
18.17	1.41	0.02	37.05	14182.46	14961.51	2770.61	664.82	15.22	0.10	718.80	
3.75	0.05	0.00	19.23	14195.85	15066.03	2797.73	668.56	15.27	0.11	738.02	
0.31	0.08	0.00	21.39	14203.13	15169.77	2831.40	669.47	15.34	0.11	759.41	

Table 8.5 Pollutant loads for rich waste

TEST CELL POLLUTANT LOADS
RICH WASTE

DAY	LEACH /	CUM LCH /	RICH WASTE LEACHATE QUALITY (mg/l)						RICH WASTE LEACHATE LOADS (			
			COD	ALK	CL	AMM	NITRATE	NI	POT	COD	ALK	CL
21	13.8	13.8	18000	3700	1600	530	9.4	0.48	2000	248.40	51.06	22.08
31	6.8	20.6	10700	4400	1400	490	2.9	0.34	1700	72.76	29.92	9.52
45	88.9	109.5	23400	5000	1600	760	0.5	0.63	2700	2080.26	444.50	142.24
70	36.1	145.6	28000	6300	1600	1200	0.03	0.15	2700	1010.80	227.43	57.76
85	117.9	263.5	24700	35600	9400	110	0.1	0.32	1950	2911.39	4196.17	1107.98
98	28.1	291.6	21100	29000	12900	9900	0.1	0.16	2500	593.75	816.06	363.01
112	170.8	462.5	13800	35800	7800	2250	0.2	0.12	1500	2357.59	6116.07	1332.55
126	70.7	533.2	11240	9500	1300	1300	0.12	0.17	2140	795.01	671.94	91.95
140	17.7	550.9	7400	9600	890	950	0.2	0.24	1980	130.83	169.73	15.74
161	17.8	568.6	4000	7500	1200	1100	0.1	0.3	1900	71.08	133.28	21.32
175	6.4	575.0	3100	5500	1100	850	0.1	0.22	1960	19.72	34.98	7.00
190	4.4	579.4	2200	6500	1300	980	20	0.22	1980	9.59	28.34	5.67
203	5.4	584.8	2100	6600	1300	1040	10.8	0.23	1900	11.36	35.71	7.03
217	4.3	589.1	2300	8500	1300	1050	0.1	0.24	1706	9.94	36.72	5.62
231	3.5	592.6	2065	6166.7	1233.3	1083.3	0.07	0.17	1735.3	7.23	21.58	4.32
239	2.7	595.3	1930.7	5000	1200	1100	0.05	0.13	1750	5.17	13.40	3.22
252	2.0	597.3	1712.5	3600	970	960	0.02	0.25	1460	3.41	7.16	1.93
273	3.4	600.7	1360	3300	1050	750	0.6	0.23	1440	4.68	11.35	3.61
287	1.8	602.5	1125	49000	1100	1020	0.3	0.2	1180	2.05	89.18	2.00
301	2.5	605.0	890	18000	900	600	0.2	0.17	980	2.19	44.28	2.21
315	3.4	608.4	720	1600	700	350	2.2	0.21	1300	2.48	5.50	2.41
343	12.6	621.1	440	2000	680	69	6.3	0.38	1200	5.56	25.28	8.60
357	26.1	647.1	500	1300	500	35	8.3	0.19	590	13.04	33.90	13.04
387	25.0	672.1	480	1900	1000	64	5	0.25	380	11.98	47.42	24.96
449	193.9	866.0	220	2797.4	1020	130	0	0.14	1330	42.66	542.42	197.78
505	177.6	1043.6	580	3000	580	270	28	0.12	760	103.01	532.80	103.01
525	146.0	1189.6	587.3	2869.5	580	263.9	14	0.12	727.1	85.75	418.95	84.68
547	85.9	1275.5	640	1930	580	220	0	0.1	490	54.98	155.79	49.82
689	81.7	1357.2	400	2160	460	67	1.4	0.04	500	32.68	176.47	37.58

RICH WASTE LEACHATE CUMULATIVE LOADS (g)										
AMM	NITRATE	NI	POT	COD	ALK	CL	AMM	NITRATE	NI	POT
7.31	0.13	0.01	27.60	248.40	51.06	22.08	7.31	0.13	0.01	27.60
3.33	0.02	0.00	11.56	321.16	80.98	31.60	10.64	0.15	0.01	39.16
67.56	0.04	0.05	240.03	2401.42	525.48	173.84	78.21	0.19	0.07	279.19
43.32	0.00	0.01	97.47	3412.22	752.91	231.60	121.53	0.20	0.07	376.66
12.97	0.01	0.04	229.85	6323.61	4949.08	1339.58	134.49	0.21	0.11	606.51
278.59	0.00	0.00	70.35	6917.36	5785.14	1702.58	413.08	0.21	0.12	676.86
384.39	0.03	0.02	256.26	9274.96	11881.21	3035.14	797.47	0.24	0.14	933.12
91.95	0.01	0.01	151.36	10089.96	12553.15	3127.05	899.42	0.25	0.15	1084.48
16.80	0.00	0.00	35.01	10200.79	12722.88	3142.82	906.21	0.26	0.15	1119.49
19.55	0.00	0.01	33.76	10271.87	12856.15	3164.14	925.76	0.26	0.16	1153.25
5.41	0.00	0.00	12.47	10291.59	12891.13	3171.14	931.17	0.26	0.16	1165.71
4.27	0.09	0.00	8.63	10301.18	12919.47	3176.31	935.44	0.35	0.16	1174.35
5.63	0.06	0.00	10.28	10312.54	12955.18	3183.84	941.06	0.40	0.16	1184.63
4.54	0.00	0.00	7.37	10322.48	12991.90	3189.46	945.60	0.40	0.16	1192.00
3.79	0.00	0.00	6.07	10329.70	13013.48	3193.77	949.39	0.40	0.16	1198.07
2.95	0.00	0.00	4.69	10334.88	13026.88	3196.99	952.34	0.40	0.16	1202.76
1.91	0.00	0.00	2.91	10338.29	13034.05	3198.92	954.25	0.40	0.16	1205.66
2.58	0.00	0.00	4.95	10342.97	13045.40	3202.53	956.83	0.41	0.16	1210.62
1.88	0.00	0.00	2.15	10345.01	13134.58	3204.53	958.69	0.41	0.17	1212.77
1.48	0.00	0.00	2.41	10347.20	13178.86	3206.75	960.18	0.41	0.17	1215.18
1.20	0.01	0.00	4.47	10349.68	13184.36	3209.16	961.37	0.42	0.17	1219.65
0.87	0.08	0.00	15.17	10355.24	13209.64	3217.75	962.24	0.50	0.17	1234.82
0.31	0.22	0.00	15.39	10368.28	13243.55	3230.79	963.15	0.71	0.18	1250.20
1.60	0.12	0.01	9.48	10380.26	13290.97	3255.75	964.75	0.84	0.18	1259.69
25.21	0.00	0.03	257.89	10422.92	13833.39	3453.53	989.96	0.84	0.21	1517.58
47.95	4.97	0.02	134.98	10525.93	14366.19	3556.54	1037.91	5.81	0.23	1652.55
38.53	2.04	0.02	106.16	10611.67	14785.13	3641.22	1076.44	7.85	0.25	1758.71
18.90	0.00	0.01	42.09	10666.65	14950.92	3691.04	1095.34	7.85	0.26	1800.80
5.44	0.11	0.00	40.85	10599.33	15127.39	3728.62	1100.81	7.97	0.26	1841.65

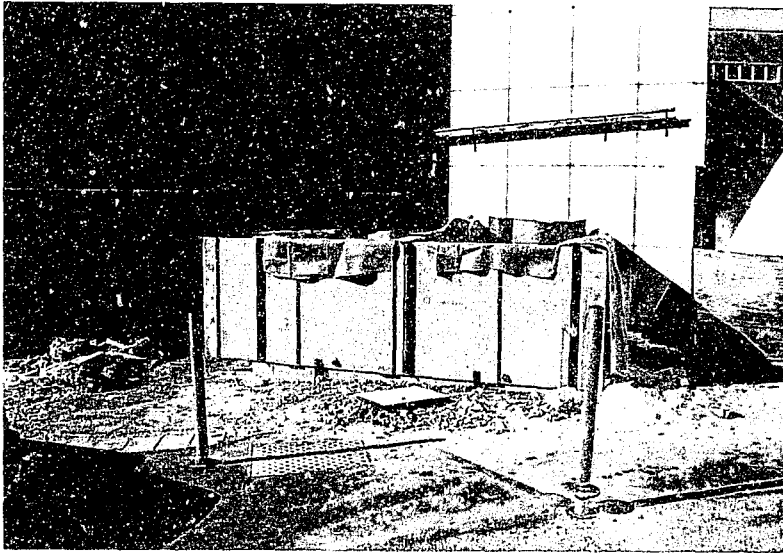


Plate 8.1 Position of test cells

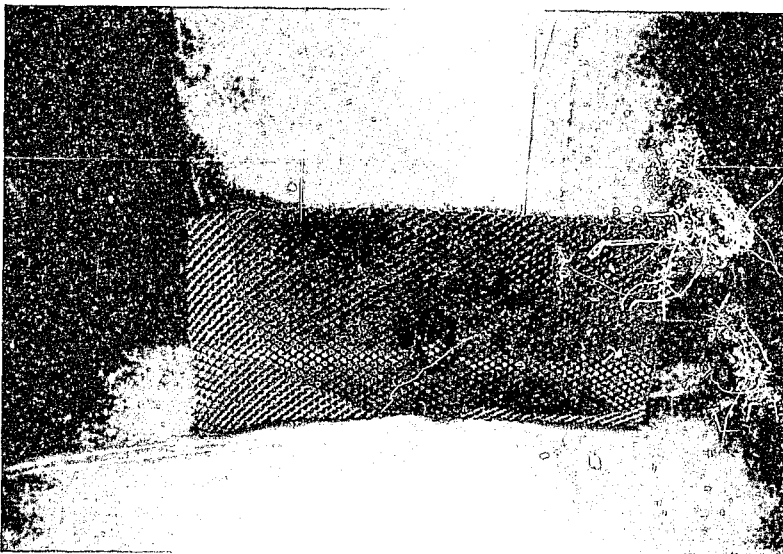


Plate 8.2 Drainage orifice protected by geomesh

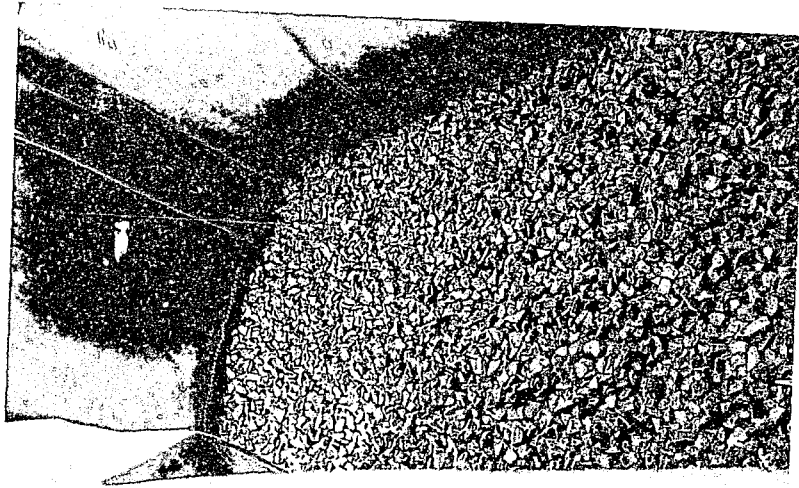


Plate 8.3 Completed drainage layer

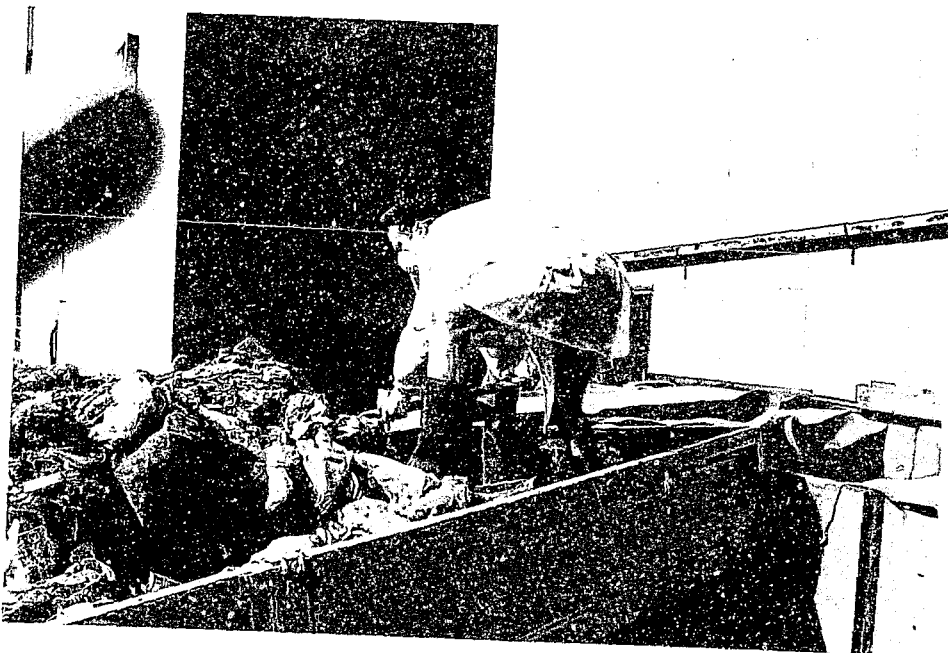


Plate 8.4 Filling of test cells

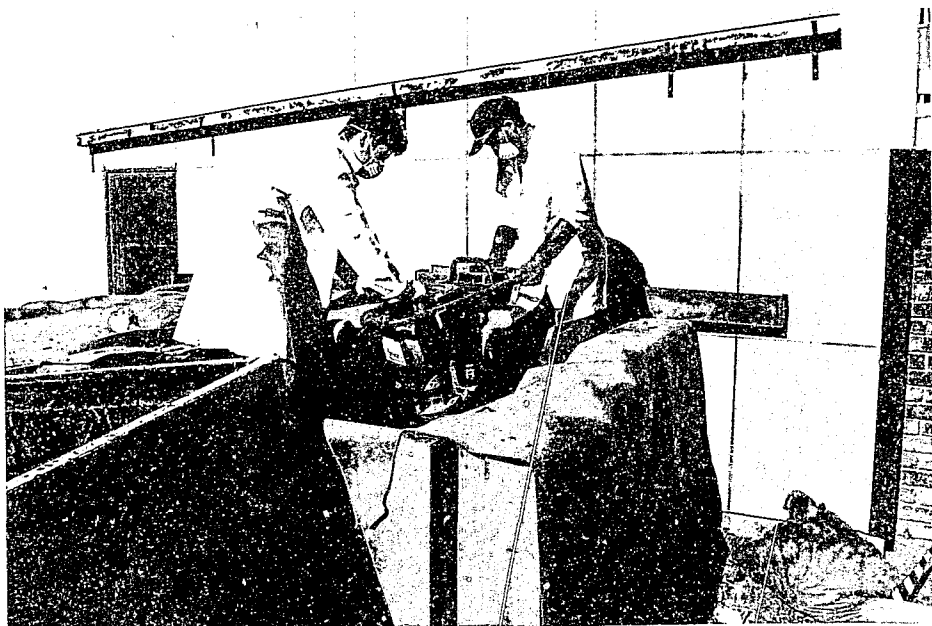


Plate 8.5 Compaction of waste using trench compactor

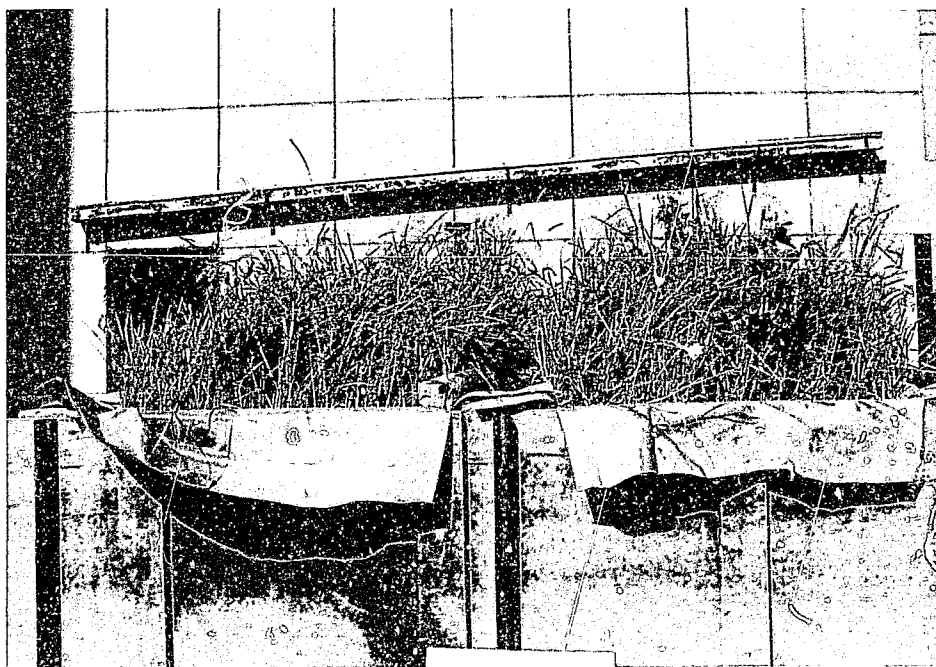


Plate 8.6 Vegetation growth in soil cover

CHAPTER 9 LYSIMETERS

9.1 AIM

As stated in chapter 6 lysimeter studies to evaluate products of decomposition are accepted practice. The full-scale lysimeter study allows the researcher to do many alternative designs, in this case different percentage compositions of waste, as well as replicate of the different cases. The small scale therefore allows more investigation to be done while at the same time keeping research costs down. The lysimeter environment can also be controlled to a greater extent, both in terms of input i.e. water and factors such as temperature. Valuable lessons learnt in the prototype study were applied to the lysimeter investigation.

9.2 STUDY METHODS

9.2.1 Lysimeter Construction

The lysimeters consisted of the same basic components as in the prototype study i.e. an airtight container, irrigation system, leachate drainage layer, gas collection system and a temperature control system. In total 18 lysimeters were constructed for the investigation.

The containers used were polyethylene drums with lids that were sealed by means of a clamping metal band. Using the results from the prototype study the smaller 50 liter drums were used. This was due to the easier handling on site as well as the limited space that was available in the laboratory.

The irrigation system again consisted of a coiled hosepipe with 1 mm holes drilled at 50 mm centres. Both ends of the hose were sealed by melting the hosepipe and then clamping it together with a set of pliers. A hole was drilled in the pipe on the opposite side to the irrigation holes and this hole inserted with a small length of tubing into a bulkhead fitting through the drum's lid. The hosepipe was then secured to the inside of the lid in a spiral shape and glued into position using a hot glue gun. In the prototype study silicone sealer had been used, however, this started showing some deterioration with age. Each lid was then linked to the modified retort stand and 2 liter coldrink bottle which served as the water delivery system. Each lid was then tested for leaks and for the rate of water delivery. Two leaks were found around the bulkhead fittings and repaired and one leak on the sealed end of the pipe. The average water delivery rate was 19 l/hr.

The gas collection system was the same as for the prototype as was the leachate collection system. All holes through the drum were made with a chassis punch and the bulkhead fittings and associated gas type fittings were constructed as shown in Figure 9.1. Steel and punched rubber washers were used to seal the brass fittings against the drum. The thread of the fittings were also sealed with white plumber's tape (PTFE). The tubing was attached to the fittings with a brass sleeve and flexing nut, the threads of which were also sealed with PTFE. The tubing was then linked either to a ball valve or to a T piece with a fixing nut. There were two bulkheads at the top of the drum, one for gas collection linked to a ball valve, and one linked by tubing to a T piece before the leachate collection ball valve.

In order to ensure airtightness O rings were manufactured for the sealing edges of the lid, one on the drum lip and one on the recessed seal area of the lid. The O rings provided a better seal when the steel clamp was attached to the drums as opposed to the existing rubber seal which did not deform under the pressure. In order to test if all the seals were working each drum was individually pressure tested. The drums

were first sealed with the steel clamp. At this stage it became apparent that, as the clamp had now to compress two O rings, it was placed under more stress than it would usually have been, with the result that a number of the clamps failed. The Department of Civil Engineering had a spot welder which was used to repair the broken clamps. Once all the drums were sealed they were inflated through the gas collection port to 200 kPa. A mixture of liquid soap and water was then poured all around the lid seal and on all the bulkhead fittings. All the pipes and valves were found to be airtight. However 7 bulkheads had to be redone and retested. All the lids had a slight leak at the join in the metal band. This problem was overcome by using silicone sealer in the seal recess at the time of final fitting. In order to test the valves, each in turn were opened while the drums were still under pressure. All the valves were working but for one where the pipe had been blocked by PTFE tape.

The leachate drainage layer was formed by a 50 mm thick layer of 25 mm washed andersite aggregate. This was the same material as used in the test cells and its inert chemical nature had been proven.

In order to stand the lysimeters in the laboratory it was necessary to create space. Two sets of lysimeter experiments were already underway in the laboratory. It was decided to mount the experiments on the wall above an existing set of drums. To carry the weight of the drums, 50 x 50 x 6 mm angle iron brackets with a square tubing compression brace were constructed. These brackets were spaced at 1 m intervals and 20 mm blockboard placed on top of the supports to act as shelving.

Due to findings with the prototype studies it was decided not to use the gas venting system on the outside of the building. The lengths of tubing required to connect the gas to the outside venting system did not allow for the free movement of the gas. It was decided to have a separate gas vent system for each drum consisting of a 2 liter coldrink bottle. The bottles were filled with a saturated solution of sodium chloride

which had hydrochloric acid added to it to bring it to a pH of 1.0. This solution was necessary in order to ensure that no atmospheric gases dissolved into it and by that route contaminate the gasses in the head space of the lysimeters.

A separate temperature control system was necessary in order to keep the drums at a stable temperature suitable for biological action. A similar system as used in the prototype study was used. The system consisted of the following components:

- A flexible heating element used in industrial processes to heat pipes was wrapped around each drum in a spiral form and secured by a length of string. The heater tapes were then connected to a 220V power source.
- In order to activate the heating elements they were connected to temperature controllers. These devices have a variable dial which allows one to set the required temperature.
- In order for the controller to function a feedback link was required. This was supplied by a temperature probe which was attached to the outside surface of the drums. One controller and one probe were used to control 3 lysimeters, i.e. 6 units for all 18 drums, this was done in order to reduce costs.
- Strips of 50 mm thick fibreglass insulation was the last component of the heating system.

All the above components were installed by an electrical contractor. The fibreglass installation was cut to size and placed along with the other temperature control equipment, to be attached once the lysimeters were filled and in place.

In order to get a temperature reading in the centre of the waste body, a copper constantan thermocouple wire was installed in each lid. A small hole was drilled through the lid and approximately 600 mm of thermocouple wire threaded through.

The wire was sealed into the lid by hot glue and tested when the drums were pressurised.

9.2.2 Lysimeter filling

The lysimeters were filled on the 5th and 6th of December 1995. The refuse used was from the Weltevreden landfill site, the rich waste being from Benoni and the poor waste from Watville. In order to eliminate scale effects and to prevent preferential flow paths developing, the waste was shredded on site. A chipper, used for reducing branches to compostable material was used. The chipper consisted of a large rotating flywheel with cutting edges onto which the waste was forced by a set of rotating spiked wheels. Although not ideal for the purpose the chipper worked well enough to reduce the waste size to pieces smaller than 50 mm. The waste was placed separately in two heaps on the back of a three ton truck, covered with a tarpaulin and taken back to the University for filling. On arrival the waste was offloaded into two separate piles and again covered with the tarpaulin. This was done so as not to affect the moisture content of the waste, either by evaporation or rain. Samples of the waste were taken and dried in a ventilated oven at 50 °C until no change in mass was observed. From this the moisture content of the wastes were found. The drums were then filled with waste. Table 9.1 illustrates the composition of different waste types that was used in each drum.

Table 9.1 Lysimeter composition and placed densities

No	Composition	Mass refuse (kg)	Total mass of sealed drum (kg)	Initial Moisture content (%dwt)	Placed density (kg/m ³)
1	Rich waste	22.36	41.44	67	559
2	Rich waste	22.84	42.00	67	571
3	Poor waste	38.9	57.22	38	973
4	Poor waste	32.66	53.62	38	817
5	50%R, 50%P	30.34	50.26		759
6	50%R, 50%P	26.52	47.38		663
7	25%R, 75%P	30.26	49.94		757
8	25%R, 75%P	29.56	49.28		739
9	75%R, 25%P	23.44	42.76		586
10	75%R, 25%P	24.06	43.22		601
11	R over P layer	30.56/22.82	48.64		746
12	R over P layer	26.14/16.66	45.54		654
13	P over R layer	32.10/10.36	51.74		802
14	P over R layer	30.02/12.06	50.62		751
15	Poor uncompacted	27.84	46.58		696
16	Poor uncompacted	30.68	47.94		767
17	50/50 tailings	30.84	51.94		771
18	50/50 tailings	27.84	51.16		696

Lysimeters 1 and 2 were filled with rich waste and lysimeters 3 and 4 were filled with poor waste. These drums were used as the controls as the same composition had been repeated in the field cells as well as in the test cells. Lysimeters 5 and 6, a 50% mixture by volume of rich and poor waste, were used as controls for the mixed cells in the field. Drums 7 and 8 and drums 9 and 10 were constructed in order to see if varying amounts of rich and poor waste had an effect on decomposition. Lysimeters 11, 12, 13 and 14 were constructed in order to determine if waste placed in layers has an effect specifically in immobilising heavy metals. As results were available for rich waste with a loose density (See Chapter 5), lysimeters 15 and 16 were filled with uncompacted poor waste. Lastly lysimeters 17 and 18 were filled with a 50% mixture of waste but this waste was covered with mine tailings, again to mirror the experiments conducted in the field.

Filling of the drums proceeded as follows. The empty drum with stone drainage layer was weighed and the scale zeroed. The shredded waste was placed into the drums in layers. These layers were then compacted with an electric hammer connected to a compacting plate. This was repeated until the drum was full. When mixtures were required it was done per spade i.e. the 75% and 25% mixture entailed mixing 3 spades of one waste to 1 spade of the other, mixing these and then compacting it. The drums were then weighed to determine the mass, and as the volume was known, the density of the waste.

Once all the drums were filled the waste was covered with a 40 mm thick layer of mixed compost and topsoil. This was used in order to simulate actual landfill conditions as well as to provide a seed, or source of natural bacteria that cause the decomposition process. The top of the soil layer was compacted until it was below the gas collection vent and pressure equalisation tube. All the cells were covered with the soil/compost mixture except cell 17 and 18 that were covered with mine

tailings. The thermocouple wire was then driven into the centre of the waste body with a steel stake.

The drum was now ready to be sealed. A bead of silicon sealer was placed around the o-ring on the drum lip and another on the inside of the lid seal. Thermocouple wire hole left by the steel stake was packed close around the wire to prevent a preferential flow path developing. The lid was then placed on the drum and the steel sealing clamp fastened. The drums were again weighed with all the components attached. This was done so that a full mass balance could be done in future i.e. to get an indication of what quantity of material leaves the lysimeter both through the leachate and gaseous emissions. All the valves on the drums were sealed at this point.

The drums were now taken to the laboratory and arranged on the shelves in numerical order. The heater tapes and temperature sensors were attached to the drums and they were then wrapped in the fibreglass insulation. At this point, the drums had already built up significant amounts of gas which became evident as the lids were bulging. The vent pipes were connected and inserted into the gas venting solution. When the gas valve was opened large quantities of gas could be seen escaping, as evidence by the vigorous bubbling in the bottles. The lysimeters were now complete and were left to decompose. The monitoring program then began.

9.2.3 Lysimeter monitoring

The monitoring program for the lysimeters set out at the beginning of the investigation was as follows:

Water addition – to simulate field conditions water, in the form of rain water that had been collected from the roof of the Civil Engineering building, was added to the

lysimeters through the irrigation system. An existing rain water trap made of geomembrane on a steel frame which fed into a storage drum was used to collect rain water which was then decanted into smaller storage containers in the laboratory. Water was added to the drums at the rate of 100 ml per week. The drums were irrigated every Friday. To reduce the gas pressure in the drums the gas vent pipe was raised to just below the top water level of the venting solution. When this was done, gas was seen to bubble off. From this we could see that the hydrostatic pressure of the solution when the gas vent pipe was at the bottom of the container kept the lysimeters at a positive pressure. If the gas was not vented off in this manner then the gas would bubble up through the irrigation system during filling which was not desirable. The explosive nature of the gas necessitated that the laboratory be made safe. This entailed an extraction fan with a sparkproof motor which was kept running at all times, sealed light fittings and locating all switches and electronic controls on the outside of the room.

Leachate – Leachate was monitored every week after irrigation. If leachate was present it would be visible in the pressure equalisation pipe. If leachate was present it was drained and sent for analysis to the Johannesburg City Council CYDNA laboratories. In order to drain the leachate off and at the same time not introduce oxygen back into the system a device was created to balance the atmospheric pressure with the pressure in the drum. If this was not done the system tended to create a suction which made draining the leachate off very difficult. The device is shown in Figure 9.2.

The following procedure was used when leachate was to be extracted:

- The valve adapter pipe was connected to a nitrogen cylinder in the laboratory and the ball valve opened.

- The opposite end of the pipe was screwed into one bulkhead fitting in the collection bottle lid. The lid was already attached to the collection bottle and sealed with a rubber gasket.
- The gas cylinder was opened and nitrogen gas allowed to fill the bottle and escape through the other bulkhead fitting. This in effect forced all other gasses out of the collection bottle.
- A drip bag which had been rolled to force all air out was connected to the free bulkhead fitting by means of a short length of pipe.
- The drip bag was allowed to inflate until it was full and the gas supply then turned off.
- The ball valve was closed and the pipe connecting the system to the cylinder removed.
- The collection bottle was then taken to the drum to be sampled and the pipe positioned at the leachate drainage valve.
- Just before the pipe was connected to the valve the ball valve on the collection system was opened and the drip bag squeezed. This forced nitrogen gas through the pipe sand into the valve thus eliminating all remaining oxygen.
- The leachate collection valve was then opened and the leachate allowed to drain freely under gravity into the bottle with all changes in pressure being absorbed by the half full drip bag.
- Both valves were then closed off and the bottle removed.
- The leachate was then transferred into a glass Schott bottle and the collection device rinsed in water ready for the next sampling. The volume of leachate was recorded at this point.

Temperature – The internal waste temperatures were taken approximately every two days. The temperatures were taken by connecting a calibrated thermocouple bridge to the length of thermocouple wire protruding from the lysimeter's lid. The active

temperature control system was not activated until later in the experiment, See 9.3 Results.

Gas – Gas samples were taken and analysed every Thursday as on Fridays all the gas was vented off for the addition of water. Leachate extraction was done on Monday mornings thus leaving three days for gasses generated by decomposition to build up in the headspace. On the days of water addition an indication of the pressure of build up gas in the drums was obtained. This was done by observing how high in the venting solution gas started to bubble out of the gas vent tube. The normal position of the vent tube was at the bottom of the bottle containing the venting solution. The vent pipe was slowly raised until the internal gas pressure was enough to force bubbles out of the vent pipe. The position of the vent pipe in the bottle was noted down.

A gas sample was taken by using the apparatus shown in Figure 9.3. Gas bombs were specially manufactured for the extraction of gas samples from lysimeters. The bombs were made from glass with a valve at one end, a large volume gas storage area and a small conical opening at the other end which could take an off the shelf syringe septum. The following procedure was used to extract a gas sample, see Figure 9.3 for a drawing of the device.

- The gas venting ball valve was closed.
- The vent pipe was disconnected and the free end sealed with Prestick to prevent atmospheric gasses from diffusing into the pipe.
- The adaptor with a piece of silicon tubing was screwed into the gas vent valve.
- The end of the gas bomb with the valve was now forced into the silicon tubing which made an airtight seal. The gas bomb has previously been aired out by opening the valve and removing the septum.

- A pipette pump was attached to the septum side of the gas bomb.
- The pipette pump's bulb was depressed.
- The gas vent ball valve was now opened and the upstream valve of the pipette pump opened, this drew gas from the drum through the bomb and into the pump.
- The pipette pump's bulb was again depressed forcing the gas out.
- This procedure was repeated 10 times, ensuring that enough gas was drawn through the bomb in order to obtain a representative sample.
- The sample of gas was now taken to gas chromatograph for analysis.

At the time the lysimeters were ready for gas analysis to be done the Department of Civil Engineering purchased a gas chromatograph, a Shimadzu GC 17A. The chromatograph connected to a desktop computer via a fibreoptic link and converter. All the data interpretation and analysis was performed by the dedicated gas chromatography software loaded onto the computer. At the time the chromatograph arrived no expertise existed within the department to operate it. The suppliers assisted in the setting up phase, the method used to analyze the gas was developed by trial and error and delivered results in a form that could be used in this dissertation. The chromatograph was set up as follows :

- Columns Molecular sieve 13 X and a Poropak Q
- Detector Thermal Conductivity Detector (TCD)
- Start time 10 Minutes
- Stop time 20 Minutes
- Carrier gas Helium
- Oven Temp 45° C
- Injection Port Temp 130° C
- Detector Temp 140° C
- Carrier Gas Flow Rate 75 ml/min

- Sensitivity Peak width 5 seconds, slope 500 per min
- Output Normalised by area

Although the chromatograph was designed to be used with capillary columns it was decided to adapt all the ports so that packed columns could be used. This was done firstly because we have experience with the packed columns from the prototype study and secondly to ensure reproducibility of the results. The molecular sieve separated out oxygen, nitrogen and methane while the Poropak column separated oxygen and nitrogen as a combined peak as well as methane and carbon dioxide. As in the prototype study the carrier gas used was helium and the detector a TCD. This combination suited the type of columns used and as they had been used before we had experience working with them. The temperature and gas flowrates used in the prototype study however could not be retained as the two chromatographs used were very different in makeup. After a series of trial and error experiments a set of temperatures and gas flowrates were chosen and used throughout the investigation.

At the injection port the temperature was set at 130° C. This ensured that when a sample was injected all the volatile components and water vapor were evaporated. This was done as these components affect the accuracy of the results as well as reducing the lifespan of the column packing and detector element. The column oven was set at 45° C and the detector at 140° C, the gas flowrate was set at 75 ml/min. The oven temperature and gas flowrate determines the separation of the peaks on the output from the chromatograph, as well as the time for all the gas peaks to come through the column. If the temperature is too high and the gas flow rate too great the analysis is over in a short space of time but the peak separation is very close or even non-existent. This means that a balance must be found between the two in order to get good peak separation and results in the shortest time.

In order to calibrate the chromatograph samples of the pure gasses (O_2 , N_2 , CO_2 , CH_4) were obtained and the same volume of gas as was injected during analysis, 0.5 ml, was injected and the time taken for the peak to emerge noted down. This was done a number of times and yielded an expected time range that if a specific column was being used at the set parameters it could be identified. The retention times were as follows :

- Poropak O_2 and N_2 0.9 min, CH_4 1.4 min and CO_2 2.3 min.
- Molecular Sieve O_2 1.8 min, N_2 3.0 min and CH_4 5.4 min.

Various methods for calibrating a quantitative analysis of a specific gas are available. The most favorable method is the use of an internal standard. This however entails the use of a standard that has all the components that you are trying to detect i.e. we would have required a standard with O_2 , N_2 , CO_2 and CH_4 in equal parts. This mixture could not be manufactured, as it was explosive so this course of action was not followed. It was decided to use the area normalisation method which states that the percentage of each component is the area under each peak divided by the total area under the curves. The Poropak column was therefore used to normalise the data while the results from the molecular sieve were used to split the O_2 and N_2 peaks on a percentage basis. A correction factor to account for the different detector responses to each component of the gas was considered but it was felt that that level of accuracy was not required for this investigation.

The gas analysis was performed by firstly filling 9 gal. bombs with gas samples from drums 1 to 9. These samples were then taken to the chromatograph which had the Poropak column attached and had been warmed up and zeroed. A 0.5 ml sample was drawn up into a "Pressure Lok" syringe and the syringe then sealed to prevent atmospheric gasses from diffusing into the sample. The sample was drawn in through a side port needle which was inserted into the gas bomb through the septum. A side

port needle was used to extend the lifespan of the septa. The needle was then inserted through the septum in the injection port, the locking device removed and the sample injected. At the same moment the record button was activated on the computer and the data recorded. When all the peaks had passed through the run was stopped and the next run started. After all 9 drums had been analyzed the columns were swapped and the Molecular sieve attached. This column had been stored in the oven with its end sealed so that it was already up to running temperature when it was attached. The system was allowed to stabilise and the Chromatograph rezeroed. Drums 1 to 9 were then analysed on the Molecular sieve. When this was complete the gas bombs were opened and allowed to vent before they were filled with gas samples from drums 10 to 18. These samples were then analysed on the Molecular sieve before the columns were again swapped and the analyses on the Poropak column carried out.

9.3 RESULTS

The course of the monitoring for this dissertation lasted 372 days. The results will be presented in the order of water balance, leachate, temperatures and gas.

9.3.1 Water Balance

9.3.1.1 Introduction

The lysimeters were irrigated once a week with 0.1 l of captured rain water. This is equivalent to a rainfall figure of 0.8 mm per week. This figure is strictly empirical and derived from previous lysimeter work conducted in the Department (Rohrs, 1995). The water balance data is presented in tabular form (see Table 9.2 at the end of the chapter) and each set of six drums presented graphically up to day 372.

9.3.1.2 Lysimeters 1 to 6

See Figure 9.4. The rain added data is included on each graph to show the cumulative amount of water added to the cumulative amount of water subtracted. As can be seen at the end of the experiment close to 5 liters of rainwater had been added to the lysimeter. At the end of the monitoring period drums 1 and 2 (rich waste only) had not produced any leachate, drums 3 and 4 (poor waste only), had produced 1.06 and 1.18 l respectively. Drums 5 and 6 (50% mixture) had produced 0.46 and 0.28 liters respectively. The leachate was only drawn off and analysed when it became visible in the pressure equalisation tube, hence the steep increments in the cumulative water balance curve. The initial starting moisture contents, as analysed over the period when the test cells and lysimeters were being filled, was 67% by dry weight for the rich waste and 38% by dry weight for the poor waste. Again using the equation from Craig 1987, the initial mass of water in the wastes were calculated. Also using the water balance equation as shown in Section 3.4 and simplifying it the following equation was used to calculate change in storage:

$$\Delta s = L - P - W_m$$

The summarised water balance components are shown in Table 9.3.

Table 9.3 Water balance including change in storage, drums 1 – 6

Drum	Initial Water (liters)	Water Added (liters)	Leachate extracted (liters)	Change in storage (liters)	Mass of waste (kg)
1	8.97	4.9	0	-13.87*	22.36
2	9.16	4.9	0	-14.06*	22.84
3	10.71	4.9	1.06	-14.55	38.9
4	8.99	4.9	1.18	-12.71	32.66
5	9.83	4.9	0.46	-14.27	30.34
6	8.57	4.9	0.28	-13.21	26.52

* No leachate detected

9.3.1.3 Drums 7 to 12

See Figure 9.5. At the end of the monitoring period all of these drums had produced leachate. The summarised water balance components are shown in Table 9.4.

Table 9.4 Water balance including change in storage, drums 7 – 12

Drum	Initial Water (liters)	Water Added (liters)	Leachate extracted (liters)	Change in storage (liters)	Mass of waste (kg)
7	8.99	4.9	0.92	-12.97	30.26
8	8.79	4.9	0.2	-13.49	29.56
9	8.38	4.9	0.15	-13.13	23.44
10	8.61	4.9	0.25	-13.26	24.06
11	9.39	4.9	0.62	-13.67	30.56
12	8.39	4.9	0.25	-13.04	26.14

9.3.1.4 Drums 13 to 18

See Figure 9.6. At the end of the experiment period all but drum 13 had produced leachate. The summarised water balance components are shown in Table 9.5.

Table 9.5 Water balance including change in storage, drums 13 – 18

Drum	Initial Water (liters)	Water Added (liters)	Leachate extracted (liters)	Change in storage (liters)	Mass of waste (kg)
13	10.14	4.9	0	-15.04*	32.10
14	9.78	4.9	0.25	-14.43	30.02
15	7.67	4.9	1.49	-11.08	27.84
16	8.45	4.9	2.09	-11.26	30.68
17	9.99	4.9	2.82	-12.07	30.84
18	9.02	4.9	0.11	-13.81	27.84

- No leachate detected

Drums 16, 17 and 18 and to a degree 13, 14 and 15 experienced operational problems which will be discussed fully in section 9.3.3. This led to the ingress of large amounts of liquid, thus rendering the water balance results for these drums unusable. For the purpose of analysis the initial moisture content of the cover applied to each drum was considered to be the same.

9.3.1.5 Conclusions

As in the test cells, leachate was first observed in the cells containing poor waste only (ignoring the data set of drums 13 to 18). Thereafter all the other drums responded with leachate production except those containing rich waste only. The change in

storage values are the amount of water that was absorbed by the waste over the course of the experiment to bring it to field capacity. It is evident that although different initial masses of waste were involved the change in storage values seem to remain reasonably constant.

9.3.2 Leachate quality

9.3.2.1 Introduction

Only two samples of leachate were taken and these were analysed at the Johannesburg's City Council CYDNA laboratory, one on day 242 and another on day 285. Again samples were analysed according to the nine key parameters established at the beginning of the investigation (See Tables 9.6 and 9.7 at the end of the chapter). The results of drums 13 through 18 will not be considered.

9.3.2.2 Leachate pH

On both sampling occasions the pH remained acidic indicating that the bacteriological activity was in the acetogenic phase. This trend is consistent throughout drums 1 to 12.

9.3.2.3 Leachate COD

Values of COD were only available on the sample taken on day 285. Of these drums 3 and 4 and drums 11 and 12 have markedly higher values of COD. Both of these sets of replicates had either poor waste only or poor waste in a layer on the base of the cell. This indicates, as in the test cells, that the initial COD values of leachate from poor waste is higher than that from a similar sized sample of rich or blended waste. Drums 5 and 6 exhibited the lowest COD's.

9.3.2.4 Leachate Alkalinity

Again drums 3 and 4 and drum 11 have markedly higher values of alkalinity, as in the results from the test cells. Drums 6 and 10 are the lowest.

9.3.2.5 Leachate Conductivity

Values of conductivity appear to be similar throughout the samples, with no marked difference between the various replicates. Slightly higher conductivity values were observed in drums 3, 4 and 11 as in the test cells. Drums 7 and 10 have the lowest conductivity values.

9.3.2.6 Leachate Chloride

Again drums 3, 4 and 11 have slightly higher values of chloride. Overall the chloride is similar to values obtained for the test cells. Drums 6 and 9 have the lowest chloride values.

9.3.2.7 Leachate Ammonia

Ammonia values are consistent throughout, which is indicative of protein breakdown. Drums 3 and 9 have the highest values while drums 10 and 8 have the lowest values.

9.3.2.8 Leachate Nitrate

Nitrate values existed for the first sampling only, with no data being available for the second sampling.

9.3.2.9 Leachate Nickel

The highest values of nickel are evident in drums 3, 5 and 12 with the lowest values being observed in drum 8.

9.3.2.10 Leachate Potassium

Potassium values are highest in drums 5 and 9 and lowest in drum 8.

9.3.2.11 Conclusions

The lysimeter data proved be valuable in that trends established in the test cells were reproduced in the lysimeters. Unfortunately more samples were needed so that a time series profile for each drum could be constructed and compared. The two grab samples analysed make it difficult to fully interpret the data and to draw some conclusions from them. It is also unfortunate that drums 13 to 18 were spoilt and that the data could not be retrieved due to contamination.

9.3.3 Temperatures

The temperature control system was designed to maintain the waste at a temperature which favoured the mesophilic bacteria i.e. 35 °C and thus optimise the refuse degradation. Initially the control system was not activated in order to establish the effect of the aerobic decomposition phase which was expected to naturally raise the waste's temperature. Thereafter, the temperature control system was to be activated and monitored so that the desired temperatures were maintained. See Figure 9.7, 9.8 and 9.9.

After the waste was initially placed, a 2°C drop occurred over the first 4 days, thereafter the waste temperatures exhibited a steady increase to approximately 2°C above the initial placed temperatures by day 34. After this a sharp drop in temperatures was observed and it was decided to activate the heating system.

The heating system was activated on day 41. It was decided to raise the temperatures in 2°C increments per day until the desired 35 °C had been obtained. On day 43 however a problem became evident on the temperature controller governing drums 16, 17 and 18. The feedback mechanism did not cut the heater tape off and these drums were raised to temperatures over 70°C. As soon as this was noticed power to the controller for 16, 17 and 18 was switched off. The next to go was the controller for 13, 14 and 15 which were heated to over 50°C. The entire heating system was then switched off until the error could be rectified. Unfortunately it was at this stage that irreparable damage was done to the last 6 drums.

Apart from the thermal shock, which it was believed the bacteria could recover from, drums 16, 17 and 18 and possibly drums 13, 14 and 15 environment was poisoned by the gas venting solution. The mixture used to vent the gas through, which was designed to stop infiltration of atmospheric gasses, consisted of a concentrated sodium chloride solution at pH 1. When the drums were heated to the high temperatures, the gas inside the drums obviously expanded. Once the heater tapes were switched off the gas then started to cool and contract. As the lysimeters were airtight, this generated a suction in the drums which then had to be equalised. The only available avenue was the gas vent port which was connected to the gas vent pipe in the solution. The lysimeters therefore drew in most of the 2 l of solution through the gas vent pipe and into the waste mass. By the time that this was detected drums 16, 17 and 18 had drawn all of their solution and drums 13, 14 and 15 had just begun. All the gas vent valves were closed off and the drums allowed to cool under suction. The suction was released with nitrogen gas and the lysimeters left to decompose with the heating system not operational although they were still insulated.

At this point the electrical contractor was called in to rectify the situation. The contractor was not cooperative blaming the equipment and not the installation. A protracted dispute followed until he came in and redid some of the wiring. All this time the drum's temperature lagged behind the ambient temperature due to the insulation. On day 166, the heating system was again activated and again drums 16, 17 and 18 were heated to over 50°C and again the system was shut down. The electrical contractor at this point refused to complete any more work. The operating lysimeters were considered too valuable to risk subjecting them to overheating and as an interim measure the heater tapes were activated for 1 hour per day to try and keep the waste temperatures up. This however had little effect and it was felt that it could be doing more harm than good by subjecting the bacteria to repeated thermal shocks. This procedure was then stopped and the drums allowed to follow ambient temperatures. As a result in mid-winter the temperatures dropped to around 8°C during a particularly cold snap.

The temperature control system was important in that it was there to optimise the biological degradation and in so doing obtain as much data in as short a time as possible. This was not possible and resulted in a delayed decomposition rate, as will be evident in the gas data. Of more important consequence was the loss of six lysimeters.

9.3.4 Gas

9.3.4.1 Introduction

The results will be presented in the order of replicate pairs so that direct comparisons can be drawn. An omission on all the graphs is the data from day 180 to day 340. At this stage the gas analysis was being done by a part time student and not by myself. The chromatograph was not set up correctly which resulted in a flame

ionizing detector being use which was not suitable for the analysis of landfill gas with the columns that were being used. Unfortunately the error only became apparent upon closer inspection of the results, and the problem was rectified on day 340.

9.3.4.2 Drums 1 and 2

See Figure 9.10 and 9.11. These lysimeters were filled with rich waste only and had both entered the acid fermentation stage but had not progressed past that to the methanogenic phase. Drum 2 was still very much in the CO₂ bloom stage while drum 1 was past the bloom stage and returning to atmospheric concentrations. These results were previously seen in the prototype study, also with rich waste only.

9.3.4.3 Drums 3 and 4

See Figures 9.12 and 9.13. Both these lysimeters were filled with poor waste only. As can be seen from the graph they entered the CO₂ bloom stage and as that started to taper off the unsteady methanogenic stage became evident. Methane concentrations then steadily rose until day 340 when carbon dioxide and methane concentrations were almost equal. This can be seen to be the start of steady methanogenesis and analysis for this dissertation was stopped at this point.

9.3.4.4 Drums 5 and 6

See Figure 9.14 and 9.15. These lysimeters were filled with a 50% mixture of rich and poor waste. These drums had also made the transition to methanogenesis with the steady increase in the concentration of methane. Drum 5 could be seen to be slightly lagging and had not reached the point of equal concentrations of methane and carbon dioxide.

9.3.4.5 Drums 7 and 8

See Figures 9.16 and 9.17. These drums can be considered to be in the unstable methanogenesis stage. They consisted of 25% rich waste and 75% poor waste blended. Methane values could be seen to be increasing and replacing carbon dioxide in the head space. At the stage of the last sample being taken however, the concentrations had still not reached the equilibrium point.

9.3.4.6 Drums 9 and 10

See Figures 9.18 and 9.19. These drums consisted of 75% rich waste and 25% poor waste blended. Again both drums could be seen to be in the unstable methanogenic phase.

9.3.4.7 Drums 11 and 12

See Figures 9.20 and 9.21. These drums consisted of a layer of rich waste over a layer of poor waste each of equal volume. These drums could be seen to be in the very early stages of unstable methanogenesis.

As described in 9.3.3 the lysimeters 13 through 18 were contaminated and the results from these were abandoned. The gas data will however be examined to establish broad trends.

9.3.4.8 Drums 13 and 14

See Figures 9.22 and 9.23. These drums are very close to stable methanogenesis being established although the CO₂ peak is short and sharp unlike the previous curves which were rounded and flat. The reason for this I believe was the ingress of oxygen

and contaminants when the drums overheated and sucked in the venting liquid. Methane concentration values however, seem to rise faster than in the previous lysimeter. These lysimeters consisted of a layer of poor waste over a layer of rich waste.

9.3.4.9 Drums 15 and 16

See Figures 9.24 and 9.25. These lysimeters were filled with poor waste at uncompacted densities. Drum 15 had a short, sharp CO₂ peak while drum 16's is low and flat. Again methanogenesis has been established in both drums.

9.3.4.10 Drums 17 and 18

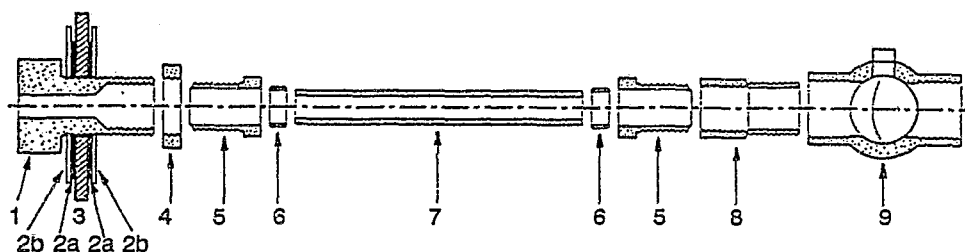
See Figures 9.26 and 9.27. These drums had a 50% mixture of waste but were covered with mine tailings. As in drums 1 and 2 the CO₂ bloom occurred and then dies off without methanogenesis being established. Unfortunately the direct cause of this, either the mine tailings cover or the poisoning by the venting liquid could not be established.

9.3.4.11 Conclusions

The gas data highlighted the difficulty in predicting when methanogenesis will be established and at what concentrations methane gas will be produced. Trends that did emerge however, was that low density organic rich waste struggles to establish methanogenic conditions in the same time period as high density organic poor but ash rich waste. From the prototype study it became apparent that on occasions methanogenic conditions are not established even over an extended monitoring period. The dense, alkali properties of high ash waste does speed up the establishment of methanogenesis even when blended with rich waste.

9.4 CONCLUSIONS

The results obtained for the lysimeter investigation did correlate in some respects with the results from the prototype studies as well as the test cells. The poor waste component generates leachate faster for the same water input but also tends to establish methanogenesis at a faster rate than the rich waste. This is due to the physical and chemical properties of the ash rich waste. The best data from a landfill management point of view was that obtained when the rich and poor waste were blended i.e. rapid methanogenesis as well as delayed onset of leachate production.



LEGEND

- | | | | |
|-----|------------------|----|-------------------------------------|
| 1. | Bulkhead fitting | 6. | Tubing sleeve |
| 2a. | Rubber gasket | 7. | 6mm ϕ silicon tubing |
| 2b. | Steel washer | 8. | Adaptor (metric to imperial thread) |
| 3. | Drum wall | 9. | Gate valve |
| 4. | Locknut | | |
| 5. | Tubing nut | | |

To connect to the gate valve, a further adaptor, tubing nut, tubing sleeve and piece of tubing are required.

Figure 9.1 Detail of bulkhead fitting

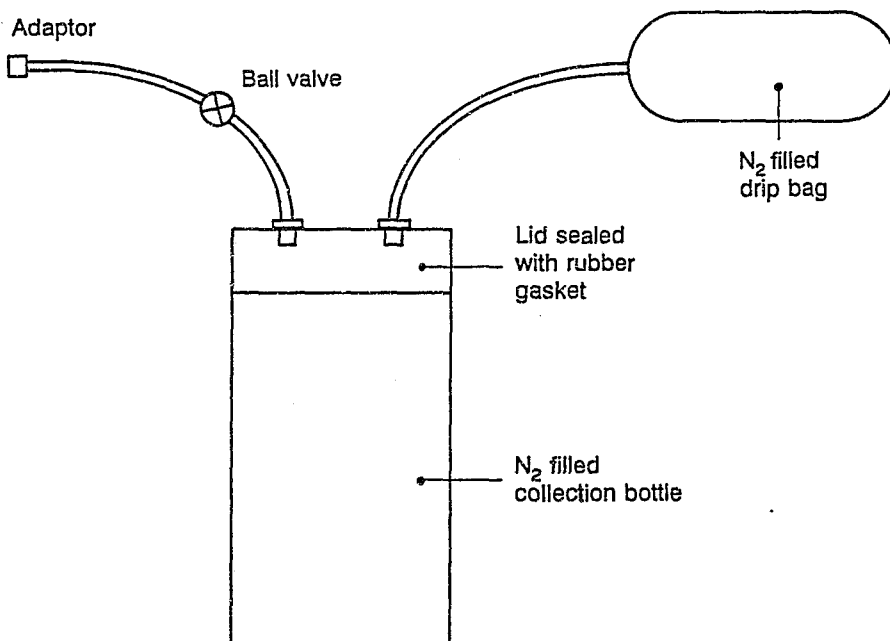


Figure 9.2 Detail of leachate collection system

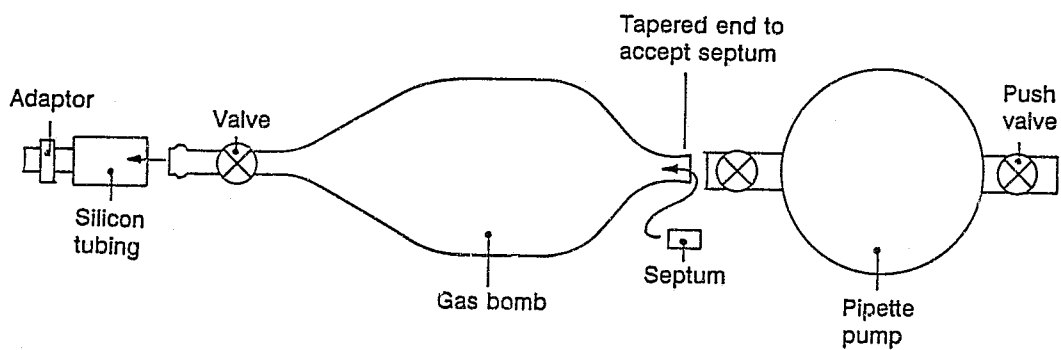


Figure 9.3 Detail of gas collection system

LYSIMETER WATER BALANCE

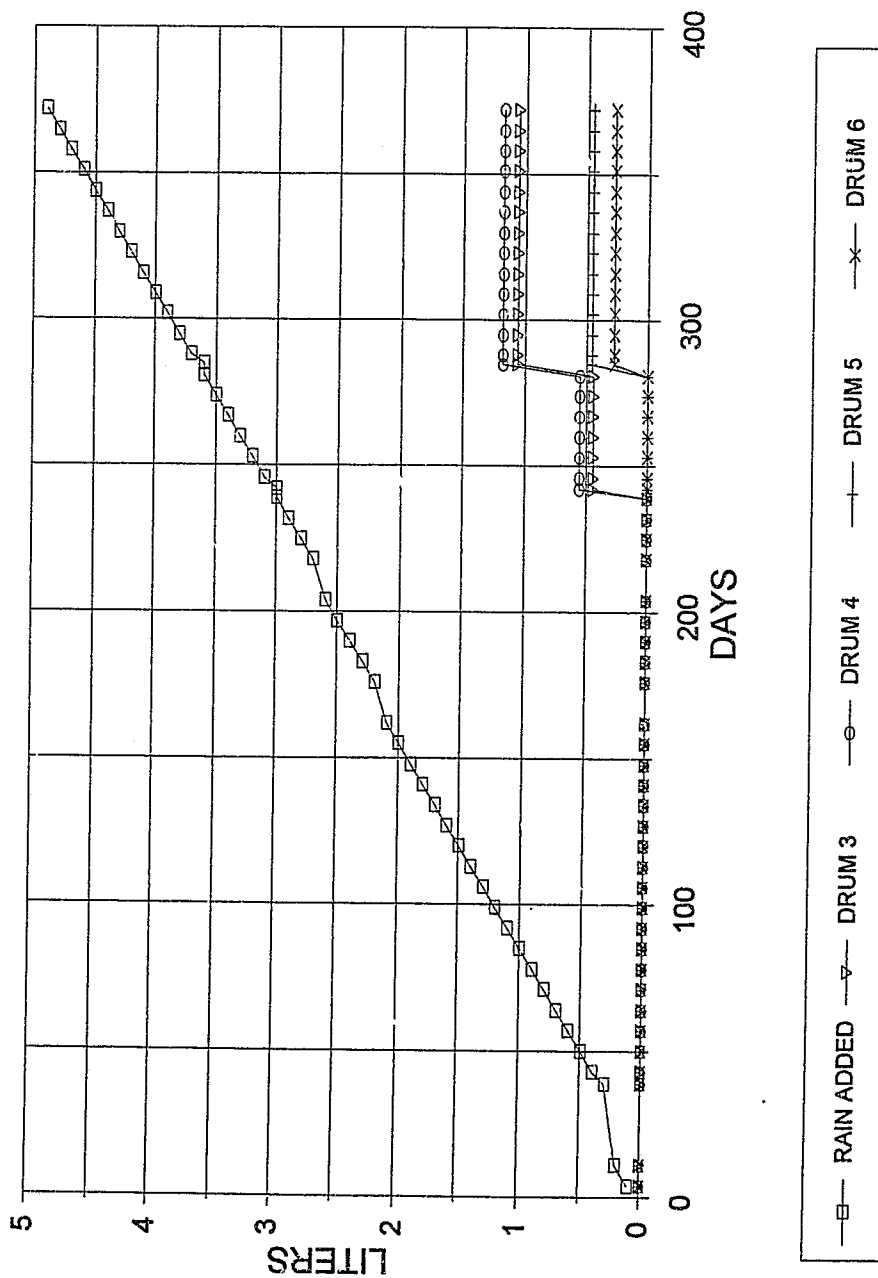


Figure 9.4 Lysimeter water balance, drums 1 - 6

LYSIMETER WATER BALANCE

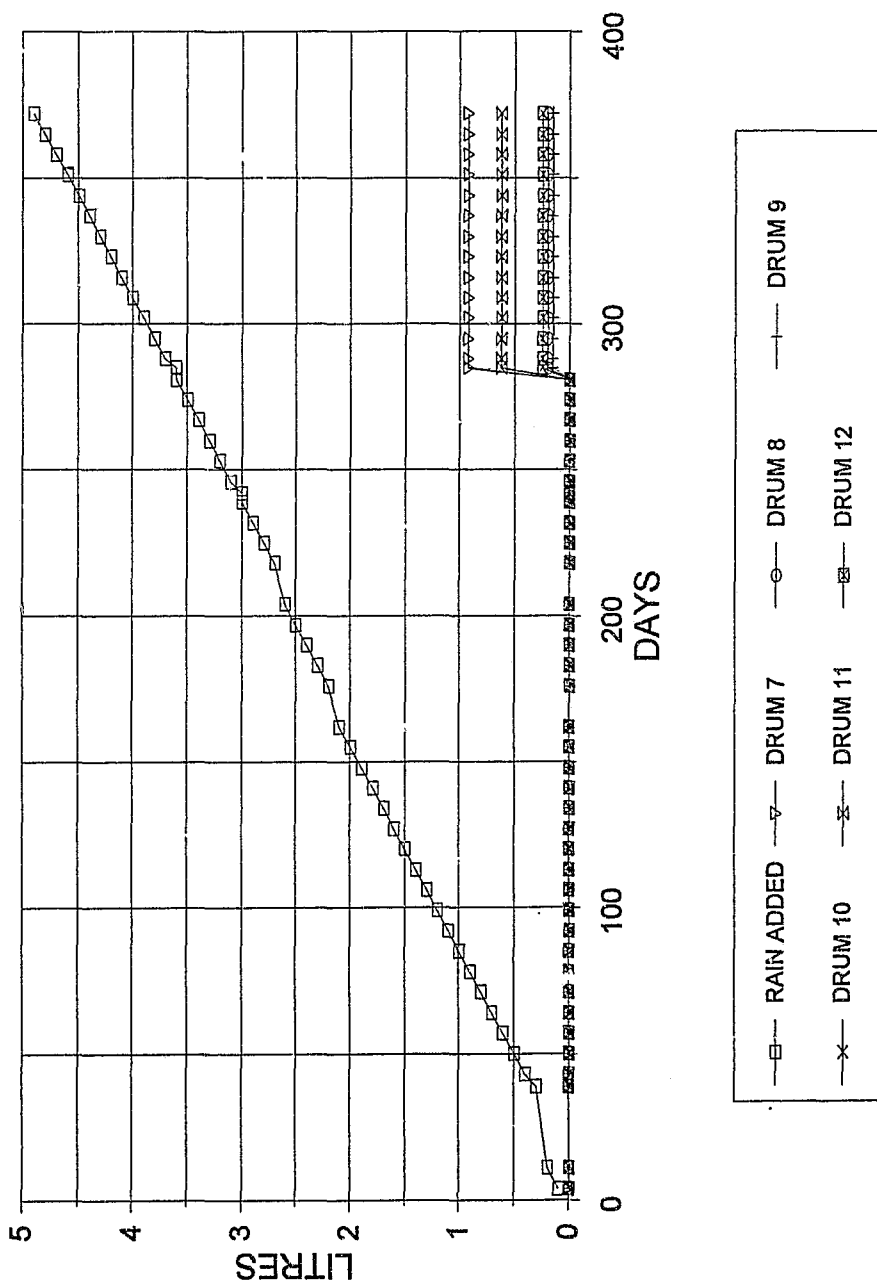


Figure 9.5 Lysimeter water balance, drums 7 - 12

LYSIMETER WATER BALANCE

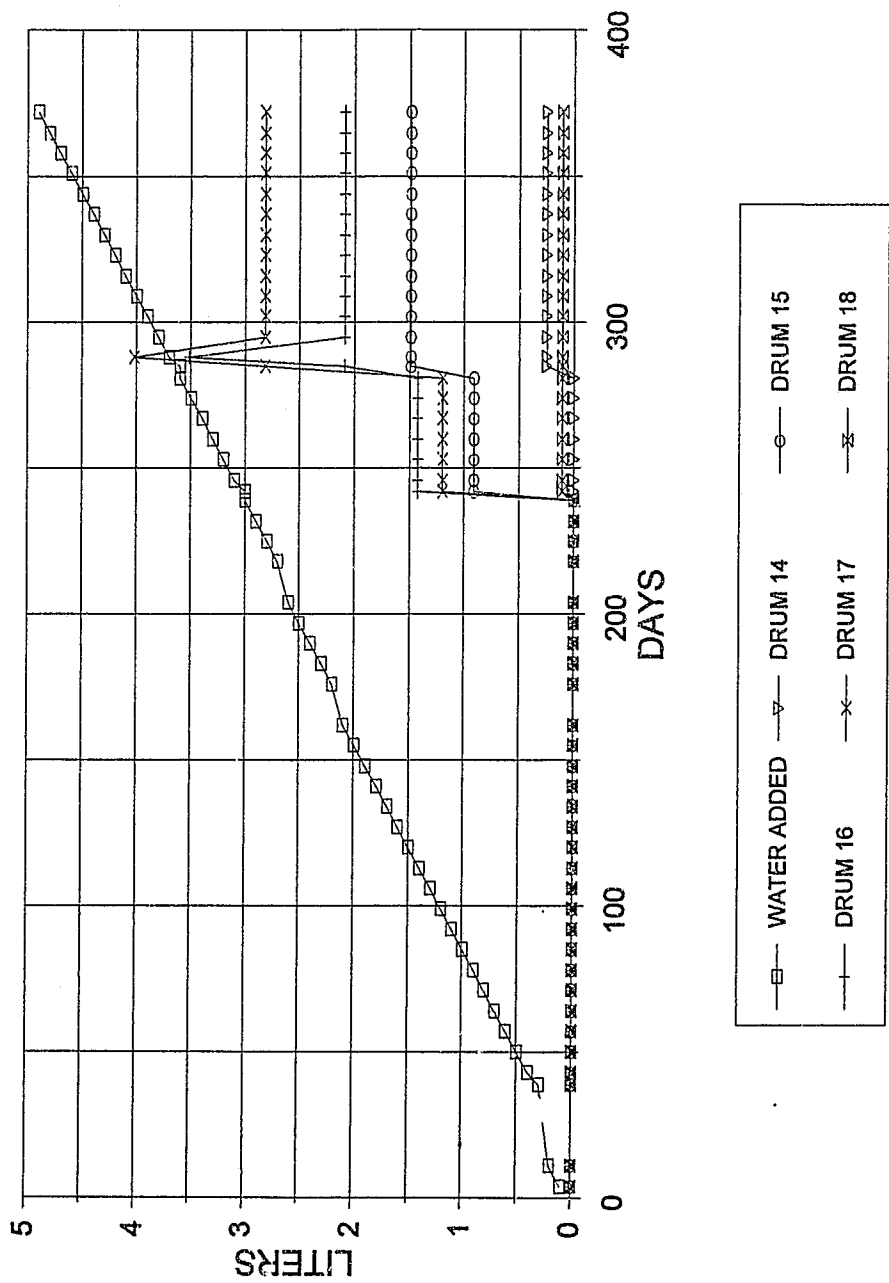


Figure 9.6 Lysimeter water balance, drums 13 - 18

LYSIMETER TEMPERATURES

DRUMS 1 TO 6

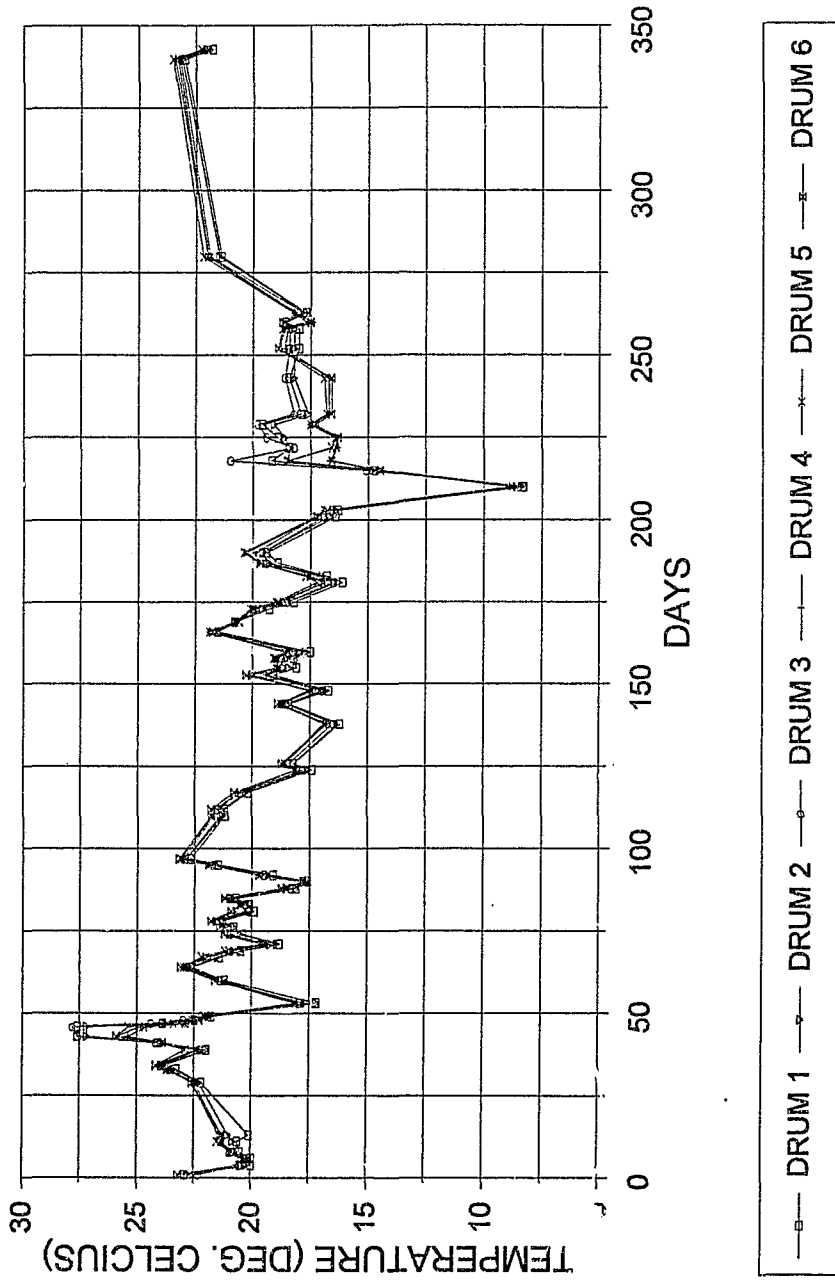


Figure 9.7 Lysimeter temperatures, drums 1 - 6

LYSIMETER TEMPERATURES

DRUMS 7 TO 12

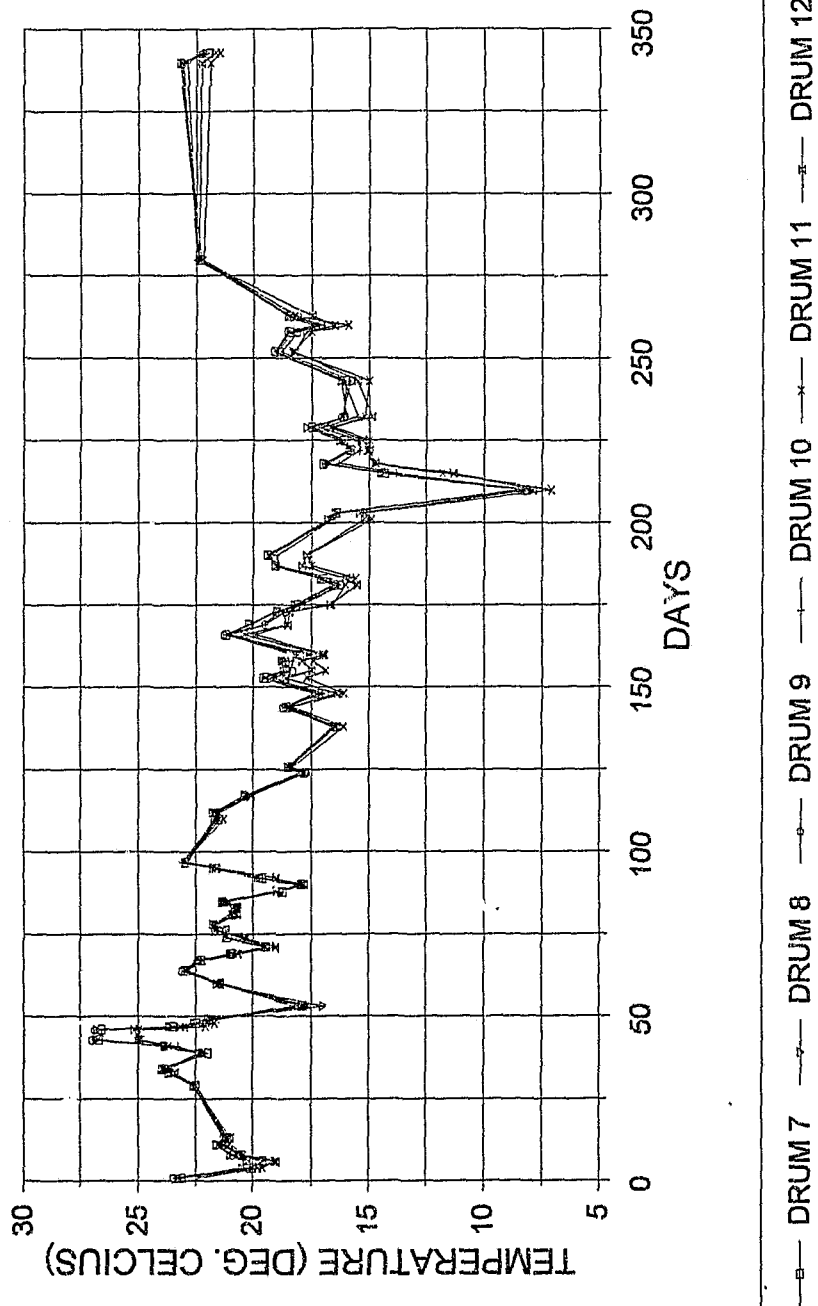


Figure 9.8 Lysimeter temperatures, drums 7 - 12

LYSIMETER TEMPERATURES

DRUMS 13 TO 18

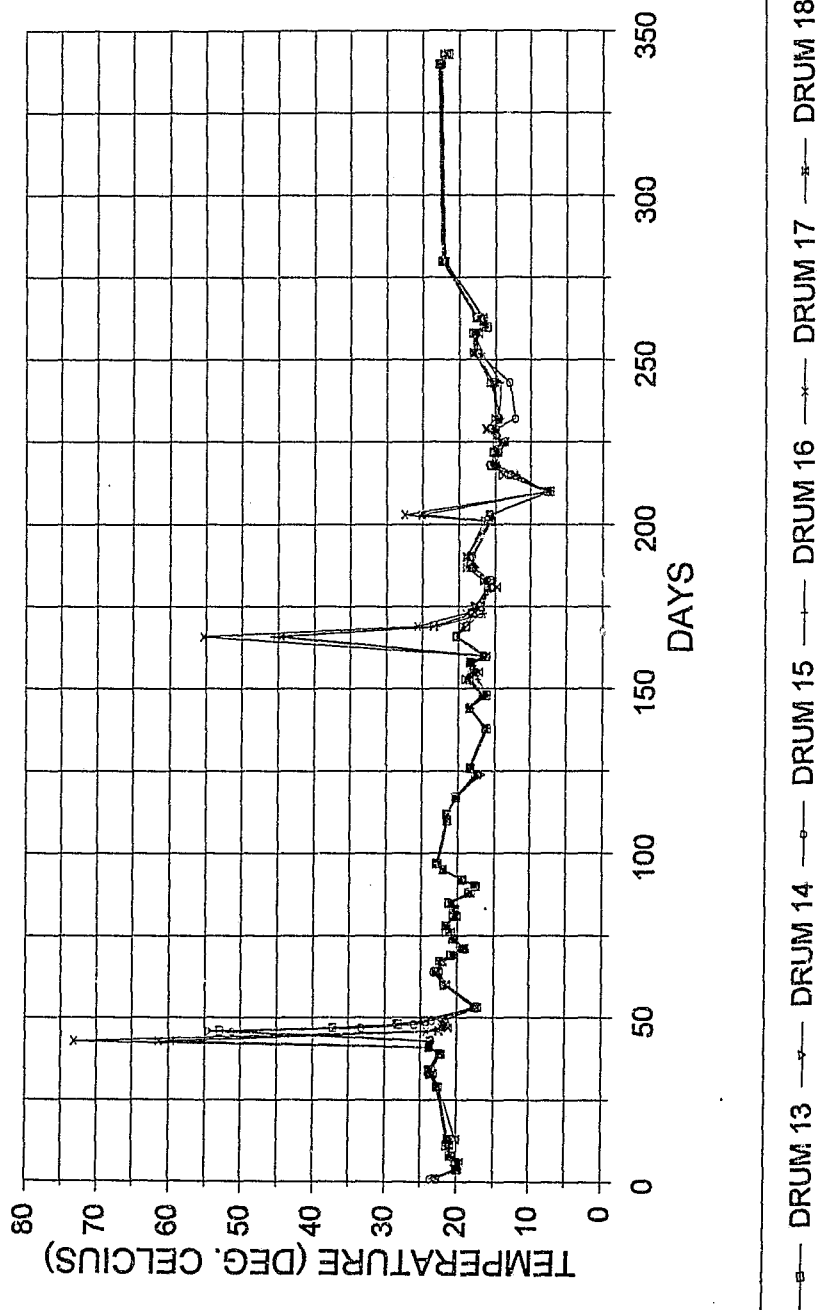


Figure 9.9 Lysimeter temperatures, drums 13 - 18

LYSIMETER GAS

DRUM 1

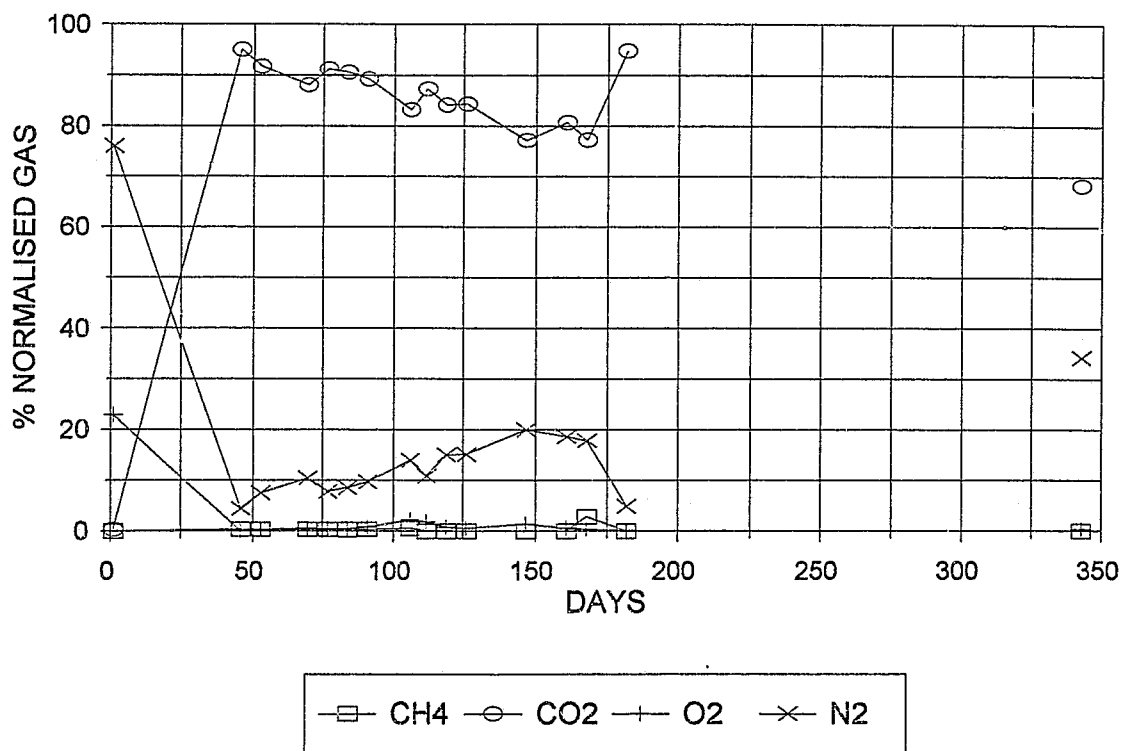


Figure 9.10 Lysimeter 1, gas analysis

LYSIMETER GAS

DRUM 2

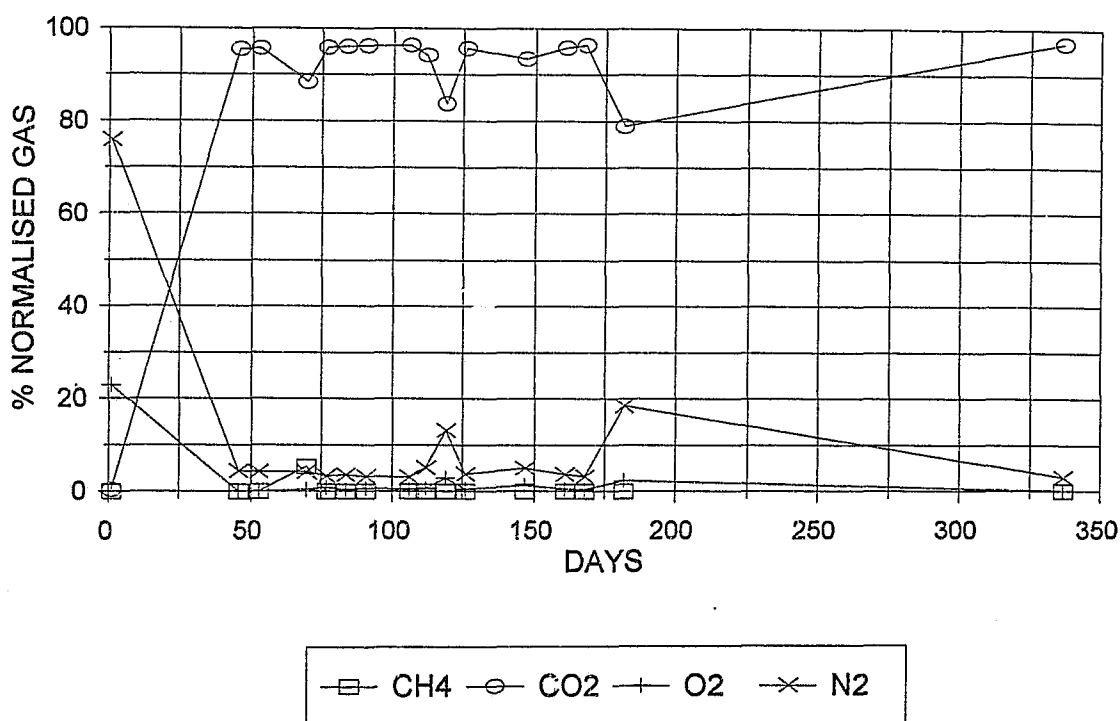


Figure 9.11 Lysimeter 2, gas analysis

LYSIMETER GAS

DRUM 3

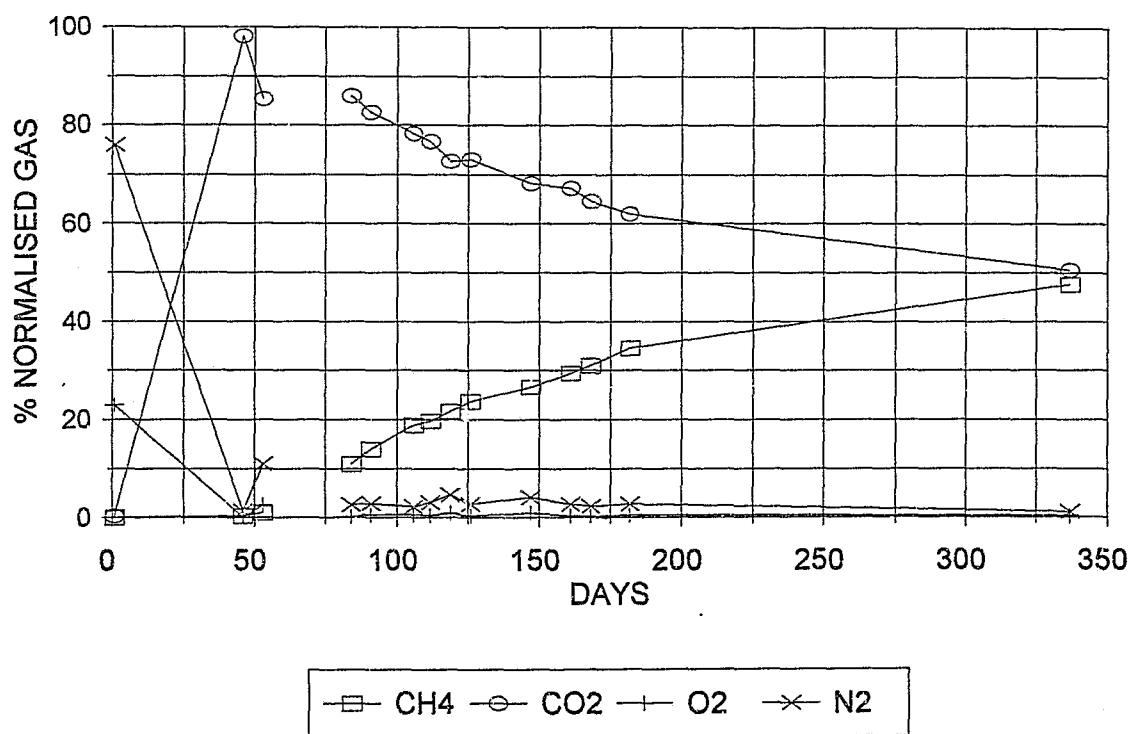


Figure 9.12 Lysimeter 3, gas analysis

LYSIMETER GAS

DRUM 4

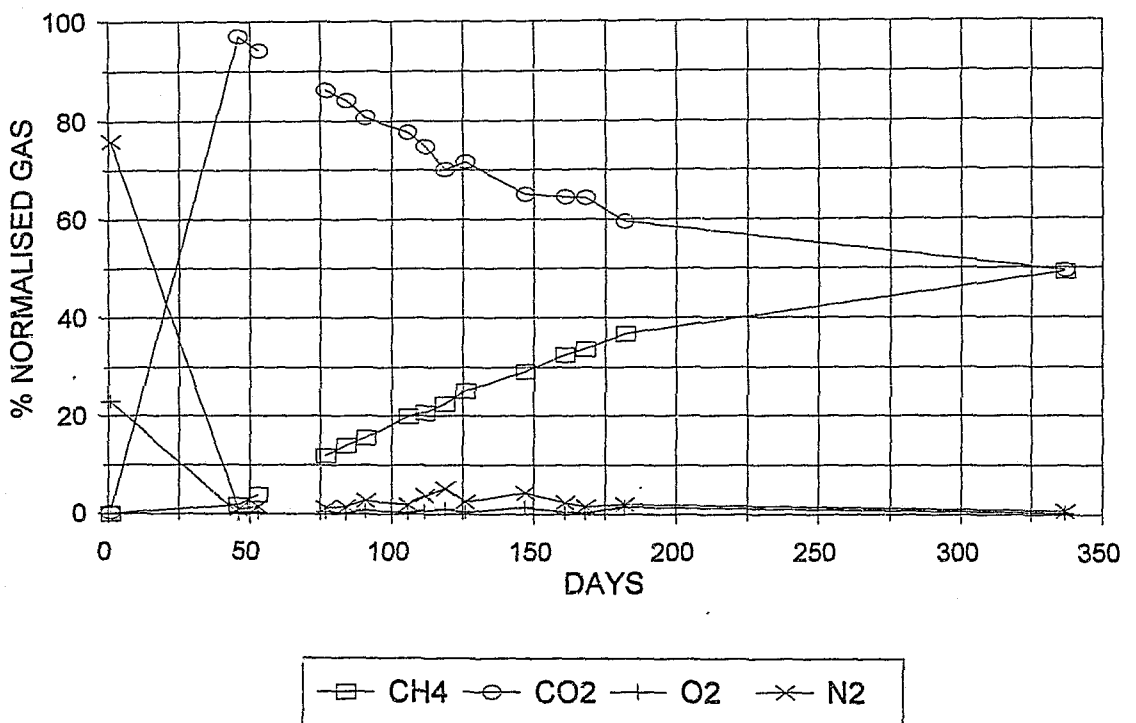


Figure 9.13 Lysimeter 4, gas analysis

LYSIMETER GAS

DRUM 5

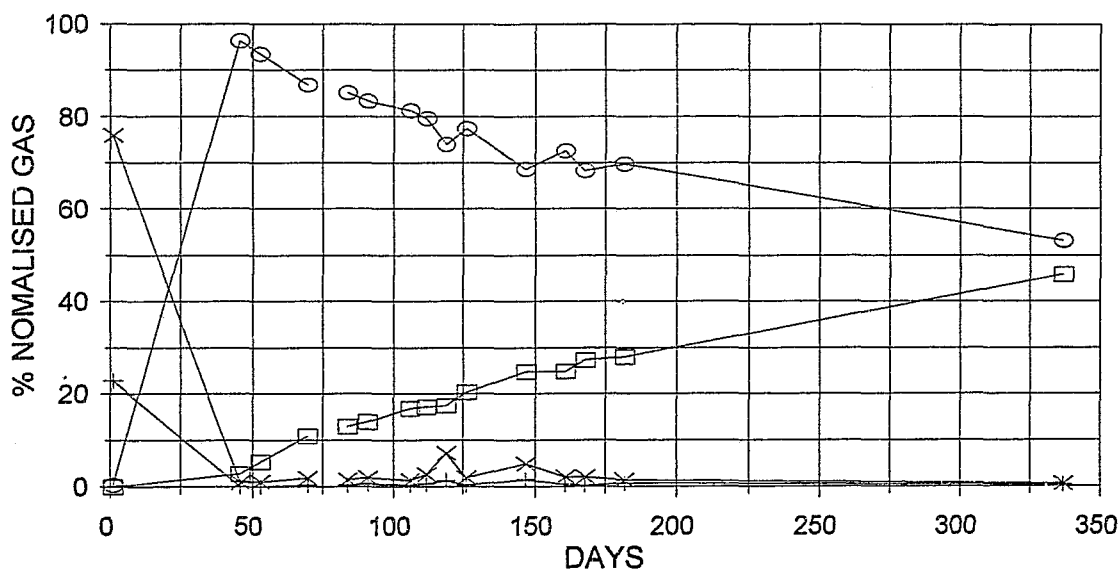


Figure 9.14 Lysimeter 5, gas analysis

LYSIMETER GAS DRUM 6

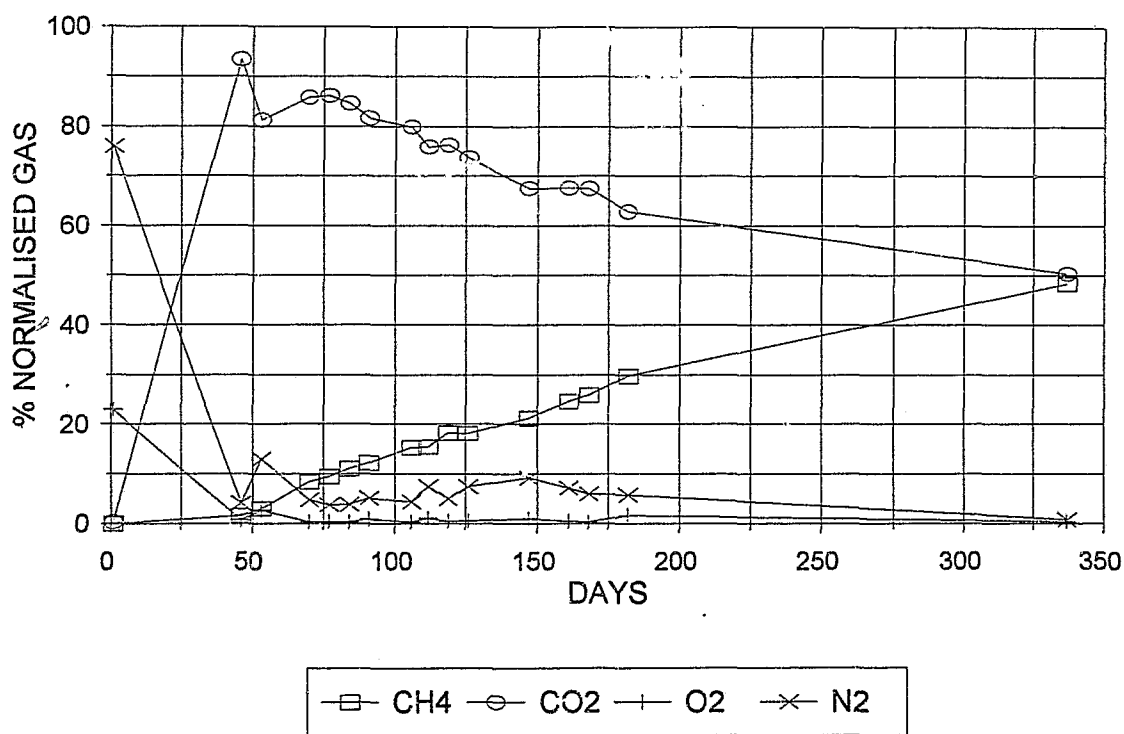


Figure 9.15 Lysimeter 6, gas analysis

LYSIMETER GAS

DRUM 7

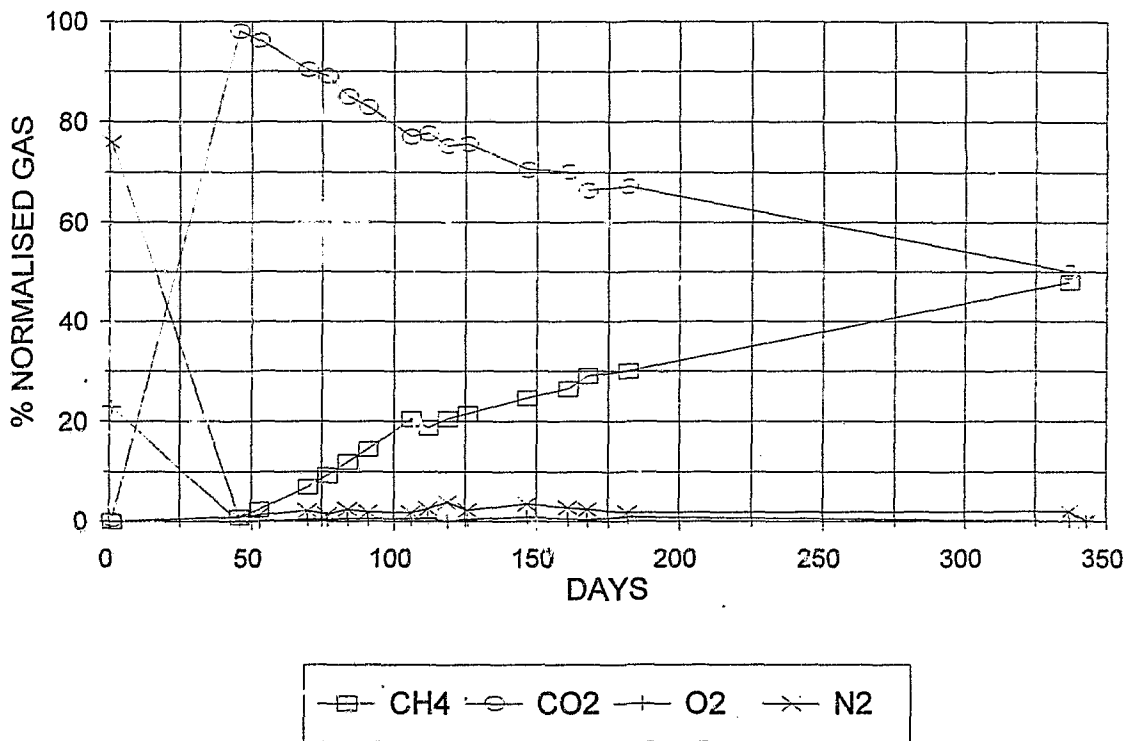


Figure 9.16 Lysimeter 7, gas analysis

LYSIMETER GAS

DRUM 8

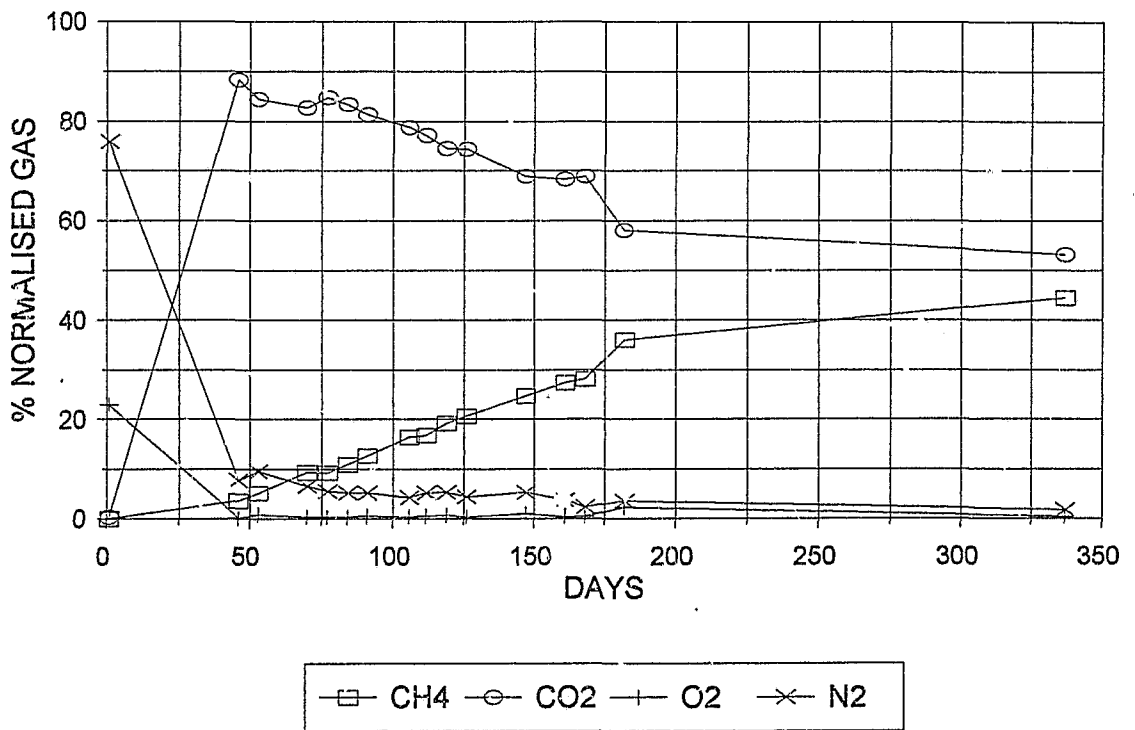


Figure 9.17 Lysimeter 8, gas analysis

LYSIMETER GAS

DRUM 9

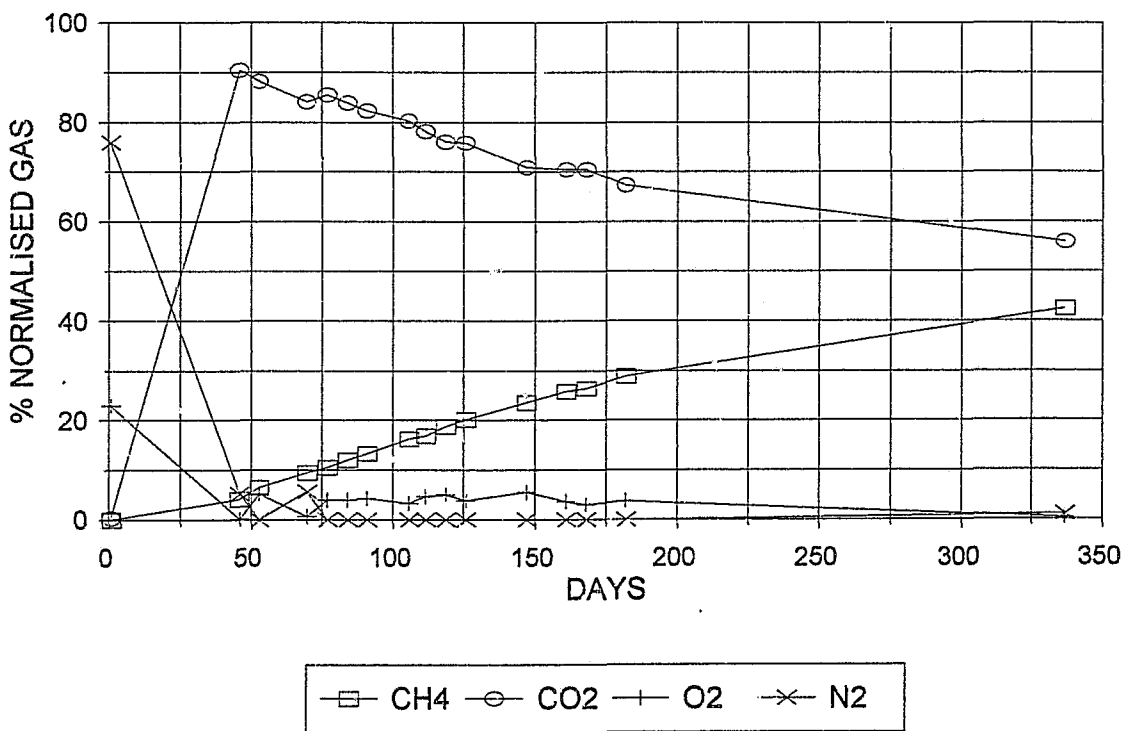


Figure 9.18 Lysimeter 9, gas analysis

LYSIMETER GAS

DRUM 10

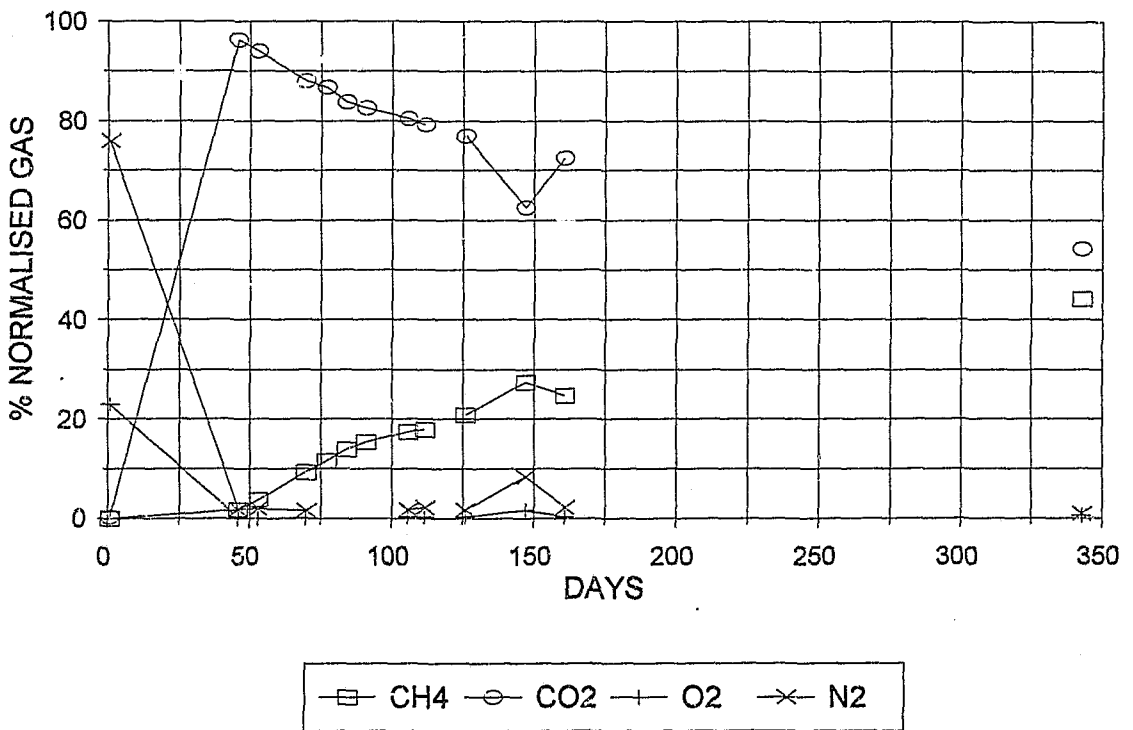


Figure 9.19 Lysimeter 10, gas analysis

LYSIMETER GAS DRUM 11

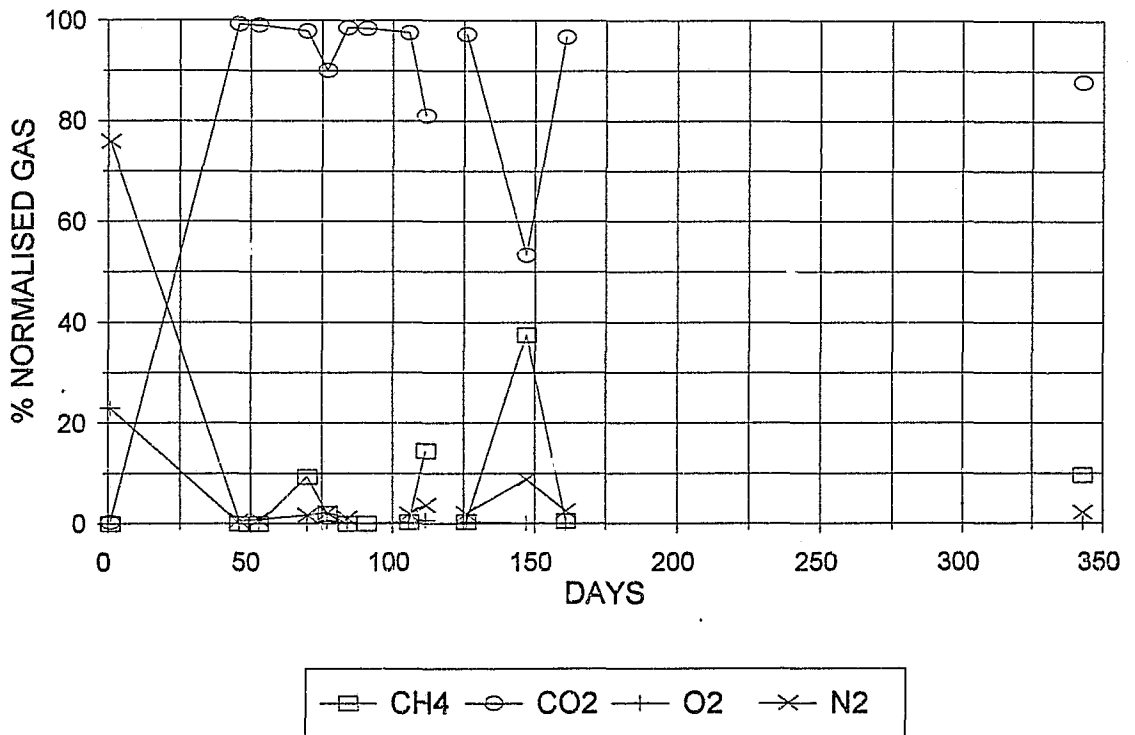


Figure 9.20 Lysimeter 11, gas analysis

LYSIMETER GAS

DRUM 12

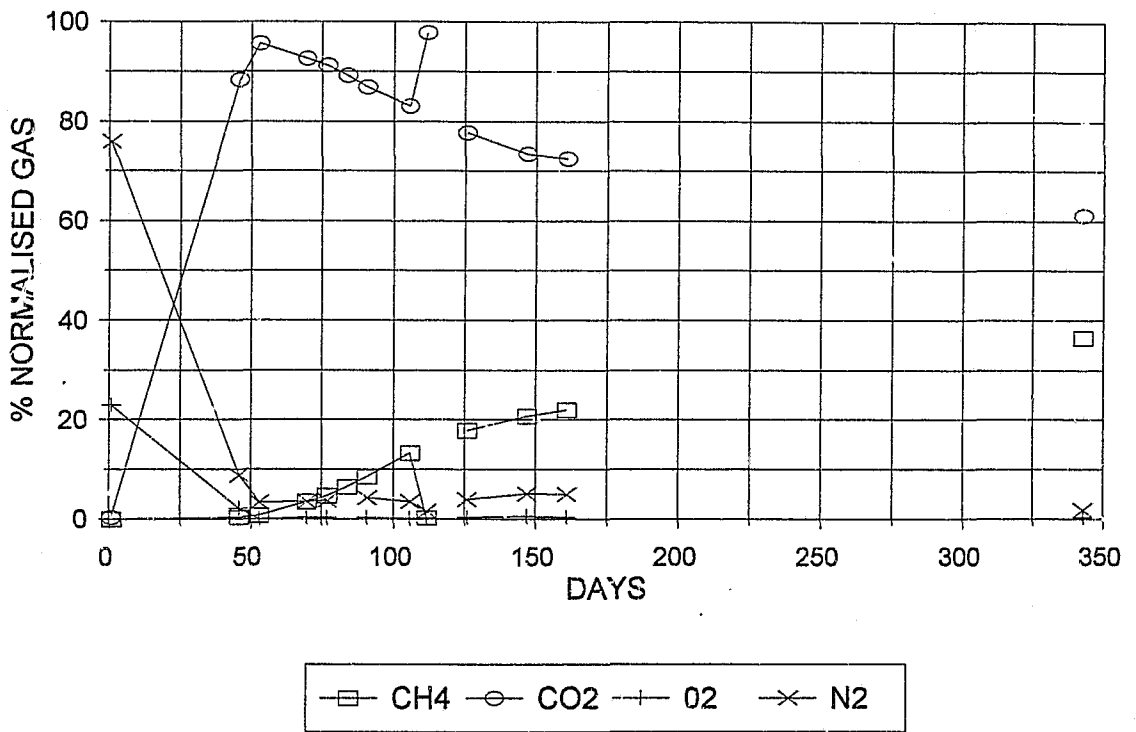


Figure 9.21 Lysimeter 12, gas analysis

LYSIMETER GAS DRUM 13

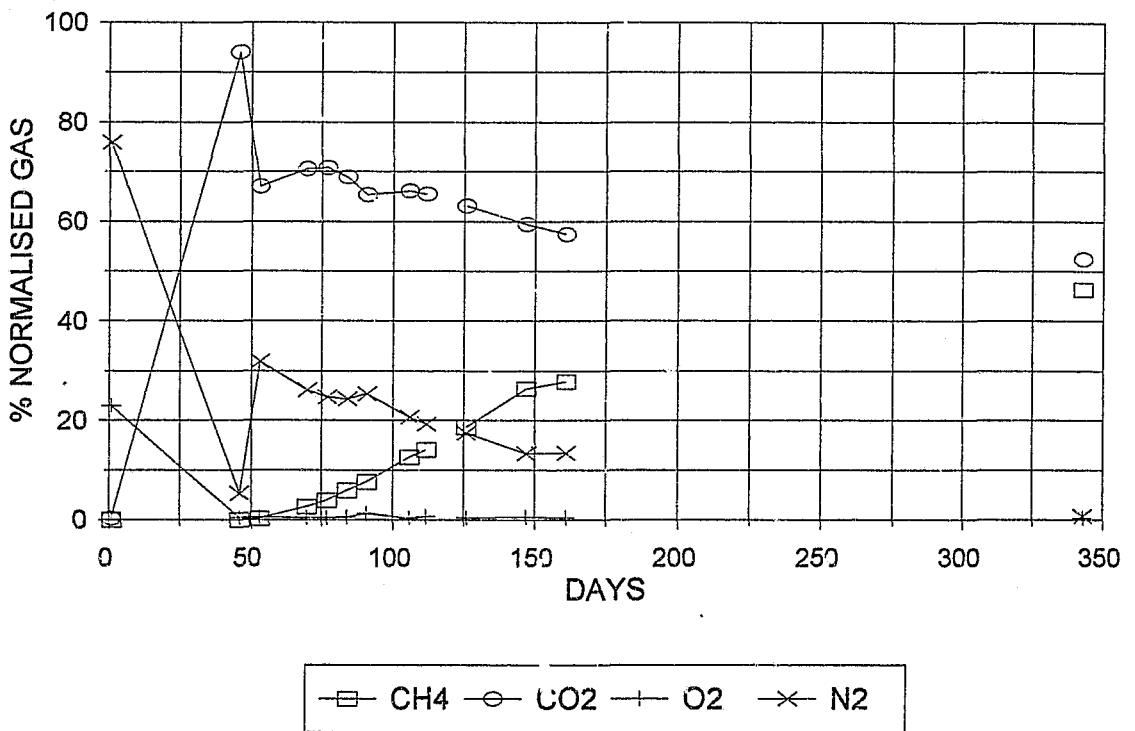


Figure 9.22 Lysimeter 13, gas analysis

LYSIMETER GAS

DRUM 14

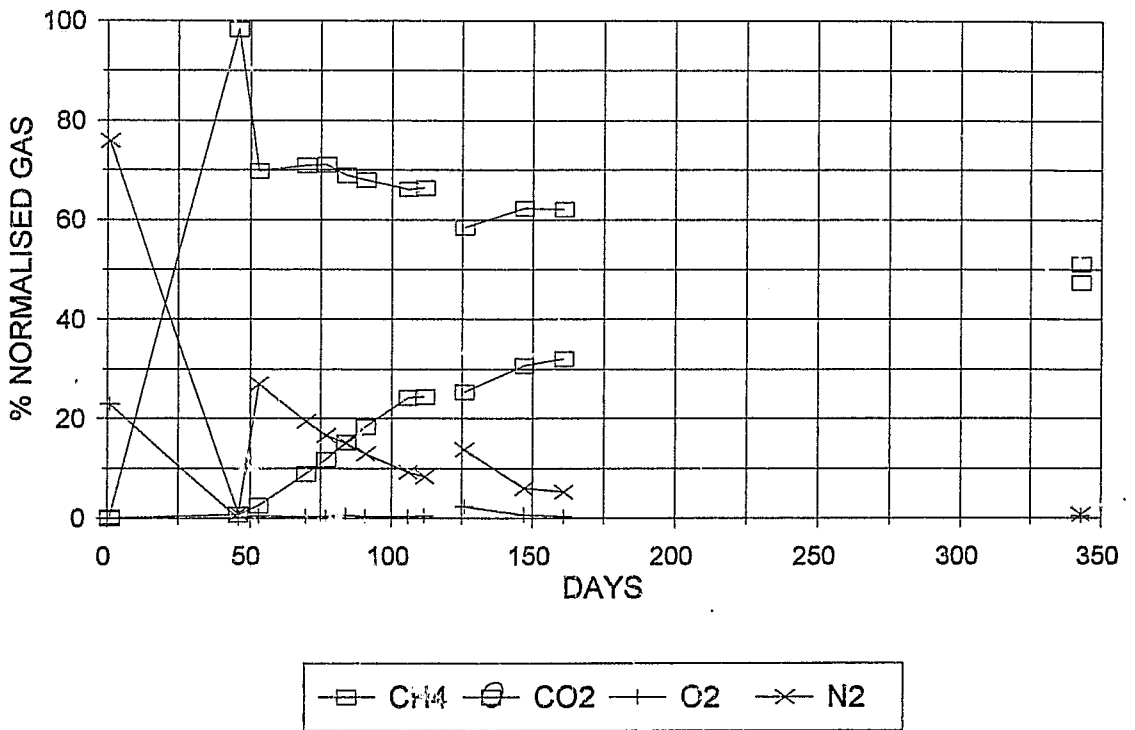


Figure 9.23 Lysimeter 14, gas analysis

LYSIMETER GAS

DRUM 15

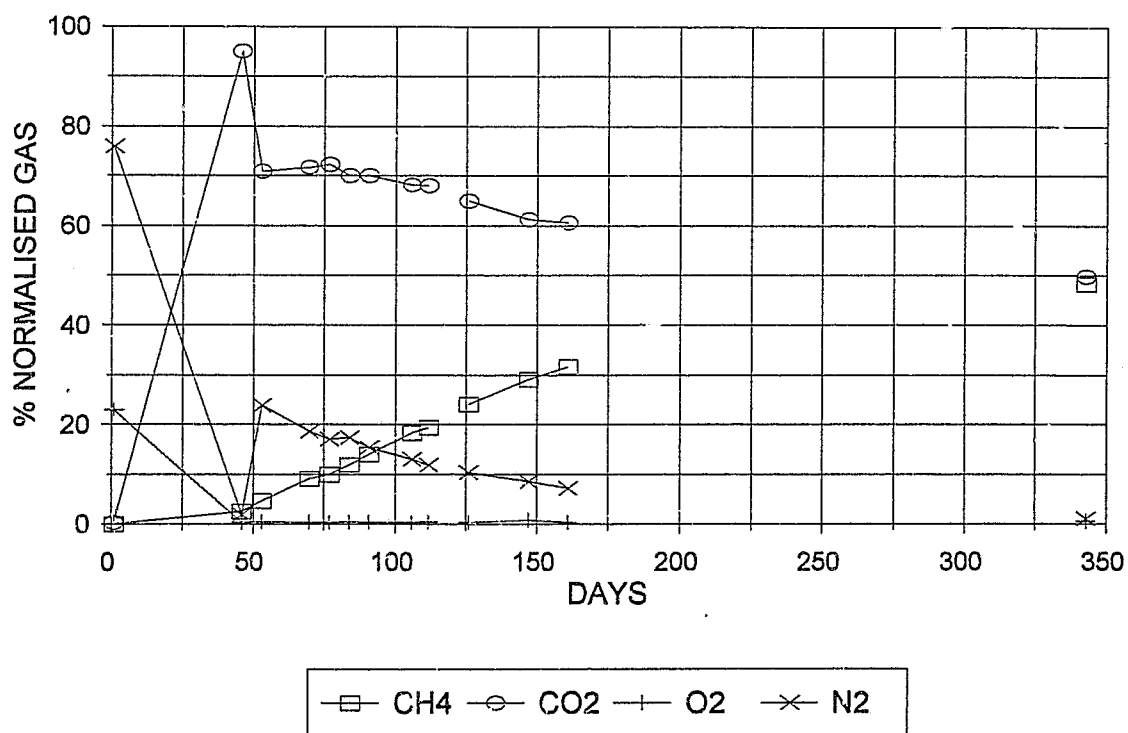


Figure 9.24 Lysimeter 15, gas analysis

LYSIMETER GAS

DRUM 16

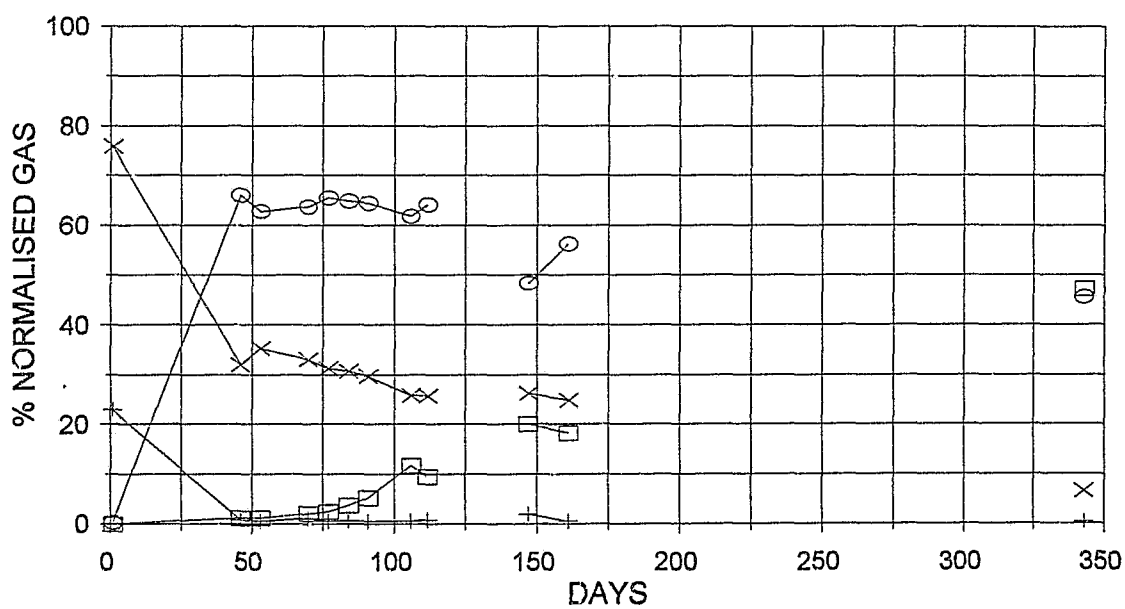


Figure 9.25 Lysimeter 16, gas analysis

LYSIMETER GAS

DRUM 17

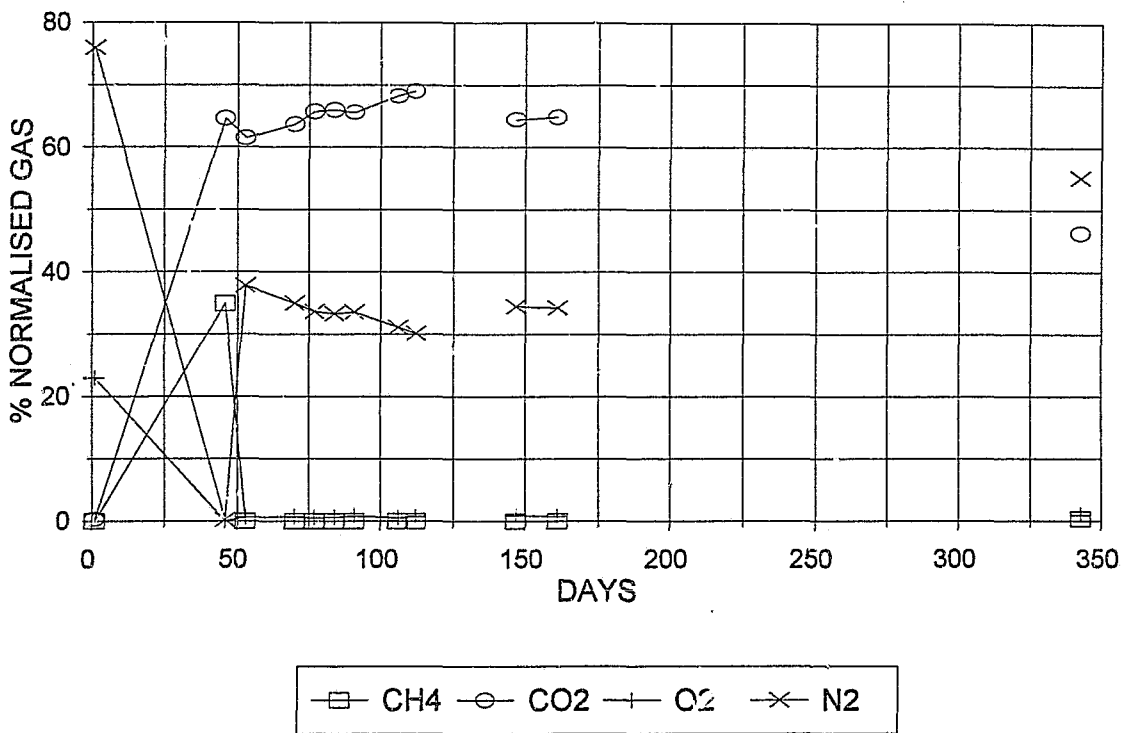


Figure 9.26 Lysimeter 17, gas analysis

LYSIMETER GAS

DRUM 18

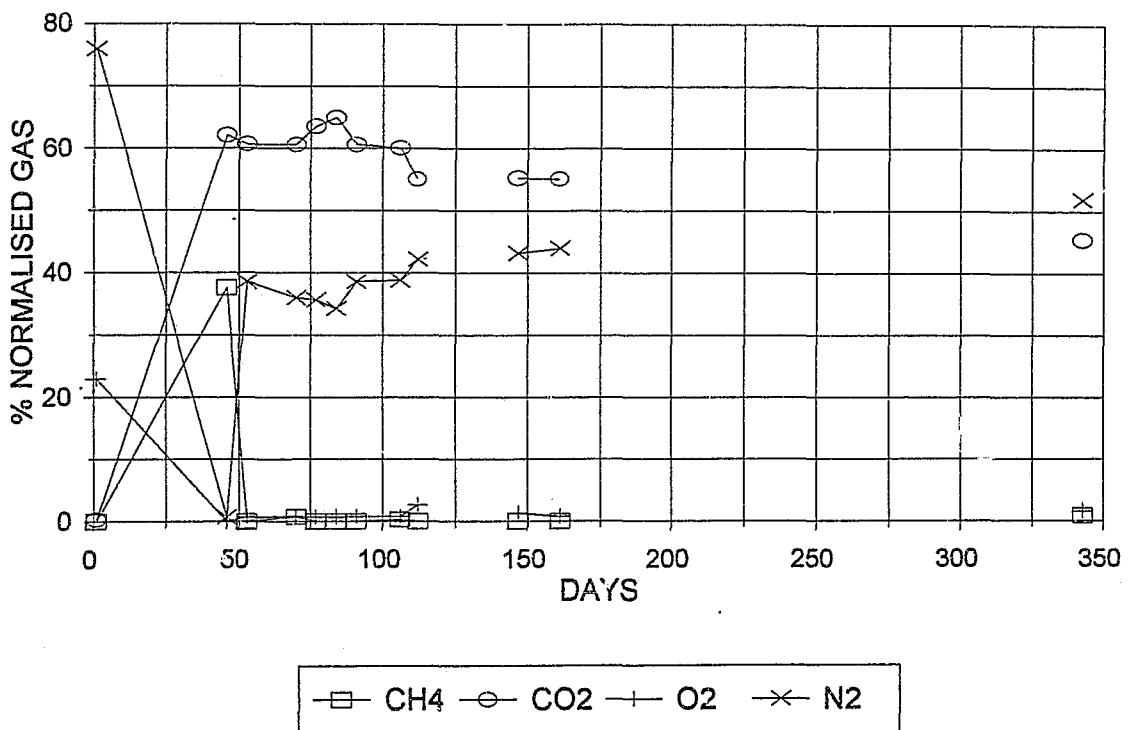


Figure 9.27 Lysimeter 18, gas analysis

Table 9.2 Water balance for lysimeters

DAY	WATER ADDED	LEACH DRUM 1	LEACH DRUM 2	LEACH DRUM 3	LEACH DRUM 4	LEACH DRUM 5	LEACH DRUM 6	LEACH DRUM 7	LEACH DRUM 8	LEACH DRUM 9	LEACH DRUM 10	LEACH DRUM 11	LEACH DRUM 12	LEACH DRUM 13	LEACH DRUM 14	LEACH DRUM 15	LEACH DRUM 16	LEACH DRUM 17	LEACH DRUM 18
41	0.1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
11	0.2	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
39	0.3	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
43	0.4	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
50	0.5	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
57	0.6	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
64	0.7	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
71	0.8	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
78	0.9	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
85	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
92	1.1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
99	1.2	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
106	1.3	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
113	1.4	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
120	1.5	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
127	1.6	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
134	1.7	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
141	1.8	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
148	1.9	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
155	2	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
162	2.1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
176	2.2	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
183	2.3	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
190	2.4	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
197	2.5	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
204	2.6	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
218	2.7	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
225	2.8	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
232	2.9	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
239	3	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
242	3	0	0	0.45	0.56	0	0	0	0	0	0	0	0	0	0	0	0	0	0
246	3.1	0	0	0.45	0.56	0	0	0	0	0	0	0	0	0	0	0.91	1.43	1.2	0.11
253	3.2	0	0	0.45	0.56	0	0	0	0	0	0	0	0	0	0	0.91	1.43	1.2	0.11
260	3.3	0	0	0.45	0.56	0	0	0	0	0	0	0	0	0	0	0.91	1.43	1.2	0.11
267	3.4	0	0	0.45	0.56	0	0	0	0	0	0	0	0	0	0	0.91	1.43	1.2	0.11
274	3.5	0	0	0.45	0.56	0	0	0	0	0	0	0	0	0	0	0.91	1.43	1.2	0.11
281	3.6	0	0	0.45	0.56	0	0	0	0	0	0	0	0	0	0	0.91	1.43	1.2	0.11
285	3.6	0	0	0.45	0.56	0	0	0	0	0	0	0	0	0	0	0.91	1.43	1.2	0.11
288	3.7	0	0	1.06	1.18	0.46	0.28	0.92	0.2	0.15	0.25	0.62	0.25	0	0.25	1.49	2.09	2.82	0.11
295	3.8	0	0	1.06	1.18	0.46	0.28	0.92	0.2	0.15	0.25	0.62	0.25	0	0.25	1.49	3.52	4.02	0.11
302	3.9	0	0	1.06	1.18	0.46	0.28	0.92	0.2	0.15	0.25	0.62	0.25	0	0.25	1.49	2.09	2.82	0.11
309	4	0	0	1.06	1.18	0.46	0.28	0.92	0.2	0.15	0.25	0.62	0.25	0	0.25	1.49	2.09	2.82	0.11
316	4.1	0	0	1.06	1.18	0.46	0.28	0.92	0.2	0.15	0.25	0.62	0.25	0	0.25	1.49	2.09	2.82	0.11
323	4.2	0	0	1.06	1.18	0.46	0.28	0.92	0.2	0.15	0.25	0.62	0.25	0	0.25	1.49	2.09	2.82	0.11
330	4.3	0	0	1.06	1.18	0.46	0.28	0.92	0.2	0.15	0.25	0.62	0.25	0	0.25	1.49	2.09	2.82	0.11
337	4.4	0	0	1.06	1.18	0.46	0.28	0.92	0.2	0.15	0.25	0.62	0.25	0	0.25	1.49	2.09	2.82	0.11
344	4.5	0	0	1.06	1.18	0.46	0.28	0.92	0.2	0.15	0.25	0.62	0.25	0	0.25	1.49	2.09	2.82	0.11

351	4.6	0	0	1.06	1.18	0.46	0.28	0.92	0.2	0.15	0.25	0.62	0.25	0	0.25	1.49	2.09	2.82	0.11
358	4.7	0	0	1.06	1.18	0.46	0.28	0.92	0.2	0.15	0.25	0.62	0.25	0	0.25	1.49	2.09	2.82	0.11
365	4.8	0	0	1.06	1.18	0.46	0.28	0.92	0.2	0.15	0.25	0.62	0.25	0	0.25	1.49	2.09	2.82	0.11
372	4.9	0	0	1.06	1.18	0.46	0.28	0.92	0.2	0.15	0.25	0.62	0.25	0	0.25	1.49	2.09	2.82	0.11

Table 9.7 Lysimeter leachate quality, day 285

DRUM	pH	COD	ALK	COND	CHLOR	AMM	NITRATINI	POI
1								
2								
3	6.06	120000	23200	3960	2500	2000		1.9 2000
4	6.25	119000	22300	3830	2900	1700		0.98 1800
5	5.87	49300	17700	3730	2200	1700		1.8 2200
6	6.28	69700	8700	3270	990	1400		0.43 1400
7	6.39	69700	17800	3200	2200	1400		0.61 1200
8	6.51	80600	13500	3320	1500	1200		0.03 960
9	6.14	70600	11800	3550	1300	1800		0.98 2200
10	5.95	48800	10000	2340	1600	940		0.67 1200
11	6.21	127000	21000	3820	2400	1500		1 1500
12	6.17	105000	15000	3740	1700	1300		1.4 1700
13								
14	5.92	71600	10700	3560	1600	1200		1.9 2300
15	6.17	69700	16800	3240	2000	1300		0.89 1200
16	6.28	64200	13600	5860	11300	1100		0.68 1000
17	6.15	70600	94000	11530	37000	790		0.94 1700
18								

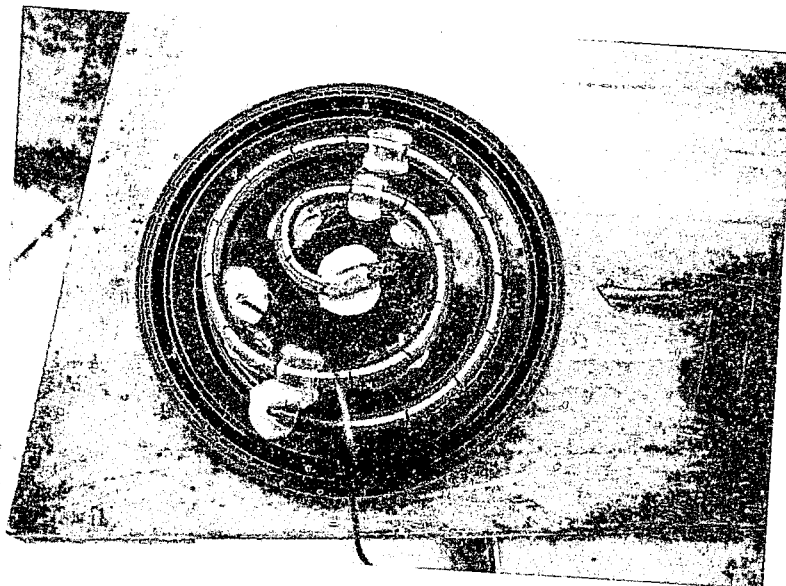


Plate 9.1 Lysimeter lid showing components

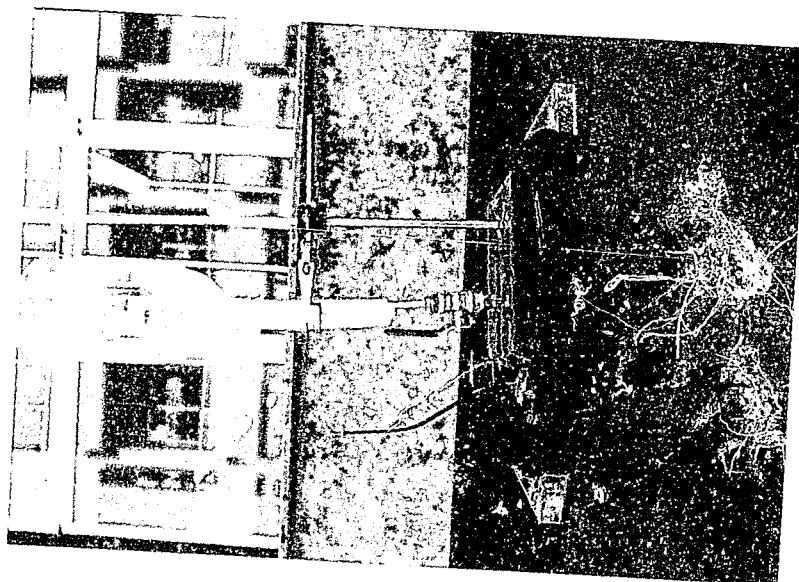


Plate 9.2 External component of irrigation system

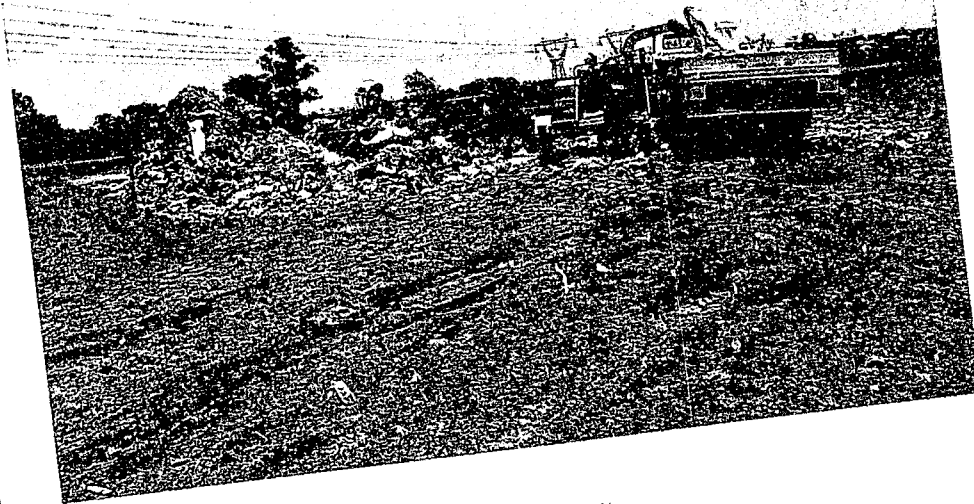


Plate 9.3 Shredding of refuse from sources at landfill site



Plate 9.4 Filling of lysimeters, note cover to prevent change in moisture



Plate 9.5 Compaction of refuse using an electric hammer

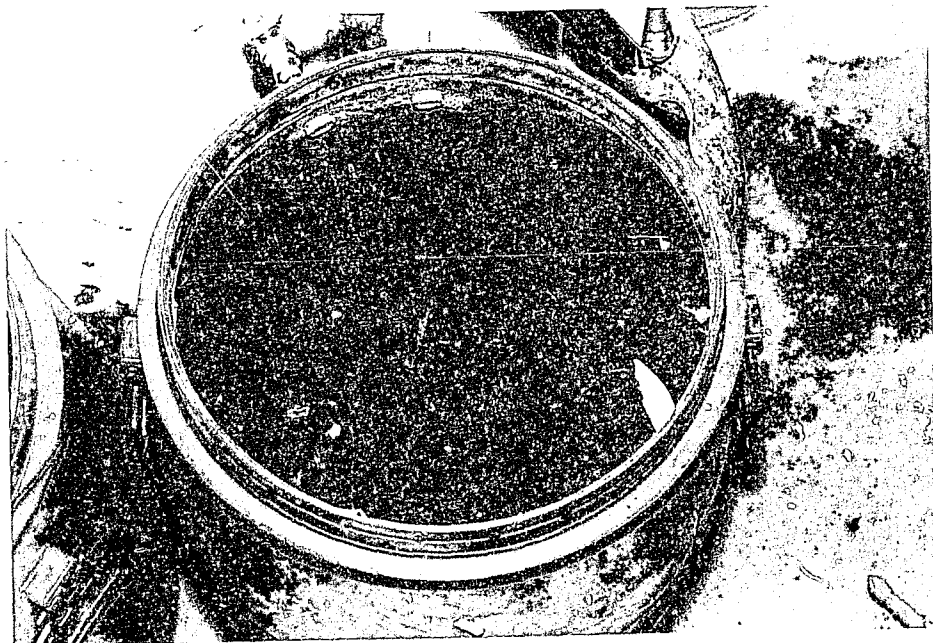


Plate 9.6 Covering of refuse with topsoil

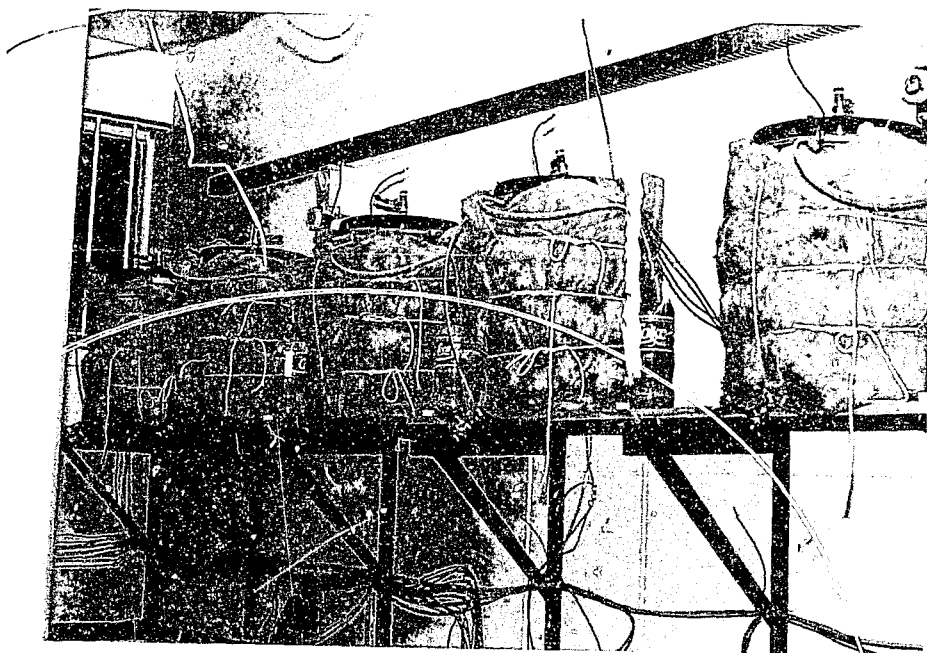


Plate 9.7 Lysimeters set up in the laboratory



Plate 9.8 Gas venting solution, notice gas bubbles

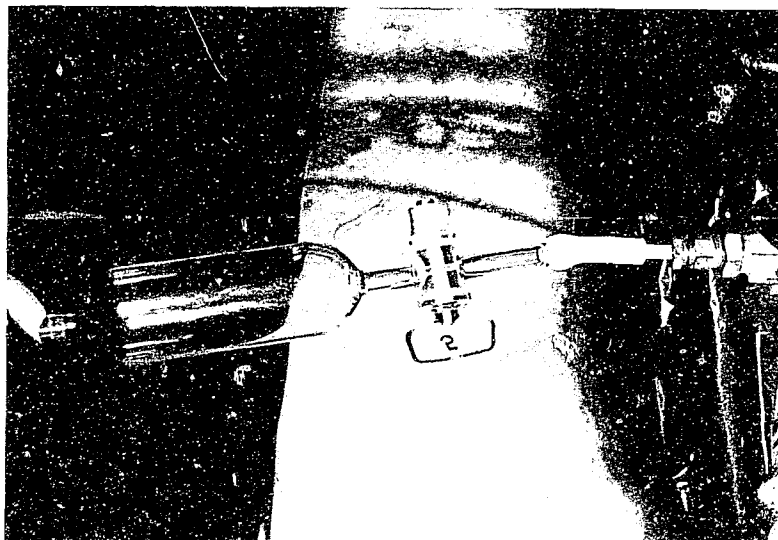


Plate 9.9 Gas collection bomb

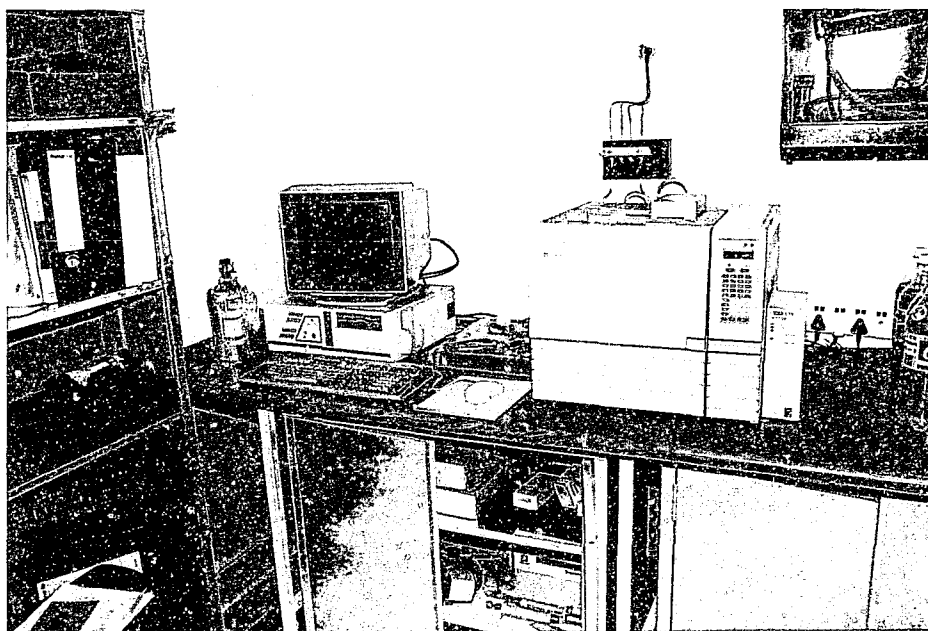


Plate 9.10 Gas chromatograph used for analysis of samples

CHAPTER 10 CONCLUSIONS

Landfill disposal represents the only technology for the ultimate disposal of wastes and is, in many developing countries, urgently needed to replace the open dump sites which can be common at the periphery of urban communities (Campbell, 1993). The development of even simple landfill practices, without the need for sophisticated site design, operation and monitoring strategies can effect a substantial reduction in adverse environmental impacts (Campbell, 1993). On the otherhand, Curi (1993) states that the standards of some developed countries on sanitary landfills are discouraging for developing countries, hence it is considered as impossible to realise the conditions stated in them.

South Africa has taken a vital step in promoting affordable waste management standards by producing the minimum requirements for waste disposal by landfill (Ball et al, 1993). Blight (1996), has stated that it may be possible to review these standards even further by considering the composition of the incoming waste. He has stated that if waste has less than 20% by dry mass of biodegradable waste it could be considered in a separate category, although these limits and the operational differences they would require remain undefined.

The characterisation study of rich and poor waste illustrated large differences in waste composition. Waste from the poor communities comprised in excess of 50% by wet weight of ash and dust, with the putrescible content never exceeding 20%. Rich waste usually had in excess of 60% by wet weight of biodegradable matter. These wastes were used to fill 4 large scale field cells, 2 test cells and 18 lysimeters, and the products of decomposition monitored.

Unfortunately no results for the field cells could be obtained. The test cells provided valuable data on the leachate production by the two types of waste. Although the

leachate from the rich waste has to date produced a leachate with a higher contaminant load than the poor waste, the differences were less than expected at the start of the investigation. If however, we analyse Table 8.6 and express the pollutant loads as grams pollutant per kilogram of original waste mass the following results are evident over the course of the experiment. See Table 10.1.

Table 10.1 Factored pollutant loads

Parameter	Unit	Pollutant load rich waste (g/kg)	Pollutant load poor waste (g/kg)	% Difference
COD	g/kg	8.36	8.07	3.6
Alkalinity	g/kg	11.82	8.62	37.1
Chloride	g/kg	2.91	1.61	80.8
Ammonia	g/kg	0.86	0.38	126.3
Nickel	mg/kg	0.2	0.06	233.3
Potassium	g/kg	1.44	0.43	234.9

As can be seen the rich waste produced higher pollutants loads expressed as g/kg for all the chosen parameters. Although the COD is not as different as was expected, this could be due to the test method and BOD should be used in follow up investigations, the inorganic and metal pollutant loads are significantly higher for the rich waste.

The lysimeter studies mainly helped clarify the gaseous emissions from the waste. It became evident that the poor waste, even when blended with rich waste proceeded to the methanogenic phase of breakdown a lot sooner than did the rich waste. This would translate to the poor waste being in the acetogenic stage for a shorter period as compared to the rich waste, which again would lessen the environmental impact of

the leachate. As the rich waste contains more biodegradable constituents it would continue to produce gas for a longer time than the poor waste. This will be confirmed by long term monitoring of the gaseous emissions from the lysimeters which will be undertaken by the Department of Civil Engineering. Unfortunately this stage could not be included in this dissertation due to the limited amount of time available. The Department will also be monitoring the long term leachate data from the test cells.

Gibbons et al (1992) suggested that due to the high ash and low organic content of poor waste, the frequency of covering on a landfill should be reinvestigated. The uncovered tip face is the main source of odours on a landfill. The low biodegradable content in poor waste would make it possible to cover the refuse on a less frequent basis. Further operational modifications would entail not having to use as much compacted effort while placing the waste, as it already fairly dense due to the dust and ash component. The loose density poor waste lysimeters went methanogenic, thus compaction to achieve methanogenesis of this waste is not necessary. Even compaction to conserve air space needs to be balanced with the cost of compacting refuse that has a loose density of around 750 kg/m^3 . Due to the low pollution loads it may be possible to locate a landfill site closer to a settlement so that it is feasible for use by low technology collection and transportation methods e.g. horse and cart or tractor trailer combinations. The requirements for site selection may also be revisited due to the lower pollution potential of poor waste.

The investigation was worth while as it confirmed firstly the difference in compositions of waste streams due to socioeconomic factors and quantified the seasonal characteristics of this variance. Secondly a clear difference in pollution potential expressed as gram pollutant per kilogram waste was found confirming that poor waste has less of an environmental impact in term of leachate quality than rich waste does. Lastly the gas data from the lysimeters confirmed that the poor waste

was likely to go methanogenic faster than the rich waste, also indicating its lower pollution potential.

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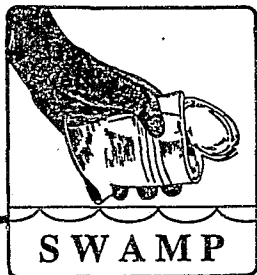
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APPENDIX A

Papers presented at conferences



THE FIRST SOLID WASTE MANAGEMENT CONFERENCE

*"Challenges for Developing
Countries"*

September 20-22, 1995
Wyndham Hotel
Kingston, Jamaica

UWICED



SOLID WASTE MANAGEMENT

"CHALLENGES FOR DEVELOPING COUNTRIES"

UNIVERSITY OF THE WEST INDIES

CENTRE FOR ENVIRONMENT AND DEVELOPMENT

in association with

THE ENVIRONMENTAL FOUNDATION OF JAMAICA

KINGSTON, JAMAICA

SEPTEMBER 20 - 22, 1995

1. INTRODUCTION

During the International Drinking Water and Sanitation Decade, proclaimed by the United Nations for 1980 to 1990, the main emphasis of work in developing countries was placed on the improvement of water supply and the safe disposal of human excreta. Solid waste management received comparatively little attention. As a result of the lack of research into this area, solid waste management is considered to be one of the most immediate and serious environmental problems confronting urban governments in developing countries and developing areas of countries with mixed economies (Schertenleib and Meyer, 1992). Solid waste management in South Africa has been orientated to servicing the needs of a minority community within the country. Literature on solid waste issues in South Africa reflects this orientation with much emphasis on sophisticated collection equipment for formal residential suburbs, and a focus on advanced disposal options such as state-of-the-art sanitary landfilling. Not much attention has been paid to the issues of solid waste management in developing communities, either here in South Africa or internationally. So far the full extent of the problems are not known and even less is known on how to address them (Blight and Mbande, 1995). There is therefore a pressing need to start to address these issues in South Africa. The challenge is to tackle social problems like poverty and rapid urbanisation while still trying to protect the environment from the effects of inadequate waste management.

2. MINIMUM REQUIREMENTS FOR WASTE DISPOSAL BY LANDFILL

The Minimum Requirements for Waste Disposal by Landfill were published in September 1994 by the Department of Water Affairs and Forestry making environmentally acceptable, yet cost-effective, waste disposal a priority in South Africa. The document recognises that the majority of South Africa is a developing country, comprising communities which cannot afford the elaborate and stringent standards prescribed in the developed world (Ball et al, 1993).

The general objective of the Minimum Requirements for Waste Disposal by Landfill is to ensure that the most cost effective means are used to protect the environment and public health from both short and long term adverse impacts of waste disposal (Ball et al, 1993). In order to achieve these objectives different minimum requirements need to be applied to different situations in a scientifically defensible way. A classification system was developed whereby landfills can be differentiated and the minimum requirements that apply to a landfill will then depend on its classification. The Minimum Requirements use three criteria in order to classify a site, namely waste type (hazardous or general), the size of the waste stream (maximum annual rate of deposition) and landfill operation and the potential for significant leachate generation and need for leachate management (climatic water balance). These three parameters are used to classify a site. The classification will then determine what needs to be done to meet the minimum requirements for site selection, permitting, investigation, environmental impact assessment, design, preparation, operation, monitoring, and closure of the landfill.

3. HYPOTHESIS

Along with the Minimum Requirements for Waste Disposal by Landfill we intend to investigate whether differentiation of waste composition with respect to socioeconomic levels can be used to further adjust the standards for waste disposal.

The critical variable affecting the composition of waste generated by a community is the socioeconomic condition of the community (Gibbons et al, 1992). Wealthy communities form "throw away" societies, whereas poorer communities have less to throw away and are more ingenious in reusing and refurbishing articles that a wealthier community would discard (Blight and Mbande, 1995). These factors, together with the methods of collection used and the use of coal and wood burning stoves for heating and cooking, results in very different waste compositions between developing and developed communities in South Africa.

Most work on refuse decomposition and subsequent leachate and gas quality and quantity has been executed in developed countries, using waste typical of that country. The waste studied therefore tends to have very high putrescible and paper components, usually well over 50% by wet weight (Farquhar and Rovers, 1973, Rees, 1980, Lee et al, 1993, Owens and Chynoweth, 1992). It is thought that the lower putrescible component in developing communities waste will produce less concentrated leachate (Blight, 1994) and therefore have less of a pollution potential.

Of greater importance than the lower organic component is the high ash and dust component. The ash results from the burning of solid fuels for cooking and heating in areas that are not electrified, although even areas that are electrified still have a significant ash component in their waste stream. The fuels burned include coal and wood, as well as components of the waste stream like paper and plastic. Work on the co-disposal of coal ash and refuse (Cossu et al, 1991) showed that the ash accelerates the biodegradation of the organic fraction of the waste and that the methanogenic phase is established sooner. If the methanogenic phase is established sooner it shortens the acid fermentation phase of initial decomposition. The initial acid phase is characterised by a low pH and a high Chemical Oxygen Demand (COD) in the leachate, whereas the methanogenic phase is characterised by a high pH and low COD, and has therefore less of a pollution potential (Farquhar and Rovers, 1973, Rees, 1980). The ash component achieves this by increasing the pH and alkalinity, thus providing a favourable environment for the establishment of methanogenic bacteria.

Rees (1980) also showed that a high density was needed to establish methanogenesis in municipal solid waste. It is important to note that even loose waste from developing communities tends to have high densities. What is unknown at this stage is the effect the ash will have on the Eh (Redox Potential) and the field capacity or moisture retention capacity of the waste, both of which are important in terms of refuse decomposition. Another factor to consider is the nutrients available to bacteria and if the ash will introduce macro and micronutrients into the waste as well as supplying a carbon source for the bacteria. Also as yet unknown, is the ability of the ash to immobilise heavy metals. Gavasci et al (1992) studied this by allowing leachate to run through natural pozzolanic pyroclastic soils and achieved favourable results, but this has yet to be proved for coal ash.

4. INVESTIGATION

The comparative study of the decomposition processes and products of first and third world refuse was divided into three main sub-problems with specific questions that need to be addressed in each.

- How does waste composition vary across varying socioeconomic levels?

This section requires the identification of waste sources typical of their respective communities and their classification.

- How does waste composition affect polluting potential?

This section comprises the majority of work and entails finding the answers to a number of important questions :

- How does composition affect refuse degradation and its variation with time?
- How does refuse composition affect leachate quality and quantity, as well as landfill gas generation?
- Is the ash component significant in decomposition as well as in heavy metal immobilisation?

- Does a different pollution potential allow for different landfilling standards?

This section will deal with how landfilling practice may be modified to make it more economically affordable to the community it serves.

5. REFUSE CLASSIFICATION

One of the critical factors identified from the start of the project was to find suitable sources of solid waste that were representative of their origins. The waste source had to fulfil a number of requirements, namely:

- Developing community waste had to be from a large informal settlement, preferably not electrified.
- Developed community waste had to be from a formal middle to upper income residential area.
- Both sources had to be in close proximity (geographically) and arrive at a central location where the wastes could be studied.
- There had to be easy access to sampling of the waste.
- Emphasis had to be placed on safety aspects.

After extensive research a source was identified at a local transfer station. This transfer station accepts waste from the town of Heidelberg as well as the town of Ratanda. Due to previous apartheid policies, the two areas are geographically very close together yet there is a vast difference in their socioeconomic levels. In a previous study conducted by Scott and DeWaal (1991), the following figures were quoted: Heidelberg supports a population of 15000 people with an average population density of 13 per hectare. The average population in Heidelberg falls into the medium income group with a small low income component. Refuse was reportedly generated at a rate of 1.6 kg/person/day at a loose density of 200

kg/m³. Ratanda supports a population of 35000 with an average population density of 118 per hectare. The average population of Ratanda falls into the sub-economic income group. Refuse was reportedly generated at a rate of 0.4 kg/person/day at a loose density of 340 kg/m³. Ratanda is characterised by two large informal settlements and relies mainly on solid fuel burning for cooking and heating. Thus the waste generated by the two towns totals 24 T/day for Heidelberg and 14 T/day for Ratanda.

5.1. METHODOLOGY

To clarify the above data, it was decided to do a classification on waste from both towns on arrival at the transfer station. From the available literature (Leroy et al, 1992, Brunner, 1986, CRC handbook, 1973), a list of 9 categories was compiled in which to differentiate the waste, namely: ferrous metals; non-ferrous metals; glass; textiles; plastics; paper; vegetable and putrescible; dust and ash and miscellaneous (including builders rubble, ceramics, unburnt coal, leather, rubber etc). After an initial period of trial and error a sampling method was developed that was not time consuming and did not interfere with the running of the transfer station.

The waste from Ratanda, which arrives loose at the station, was collected into two to three 35 l buckets. This gave a mass of approximately 20 to 30 kg for each sampling run. The waste was sieved through a 15.8 mm sieve into the dust and ash container, the rest being hand sorted into the other 8 categories. The choice of the 15.8 mm sieve was to minimize sorting time, although nearly 80% of ash passing the 15.8 mm sieve passed through the 4.75 mm sieve as well, the balance being mainly unburnt coal, feathers and cigarette ends. The buckets with the different categories were then weighed and the composition determined on a wet weight basis. The Heidelberg waste arrived in refuse bags in a 19 m³ compacter. It was decided to classify three to four 85 l bags of refuse per load. Only unbroken bags were chosen for analysis. The bags were taken to the sorting area, broken open and their contents sorted. One of the main objectives of the classification was to determine how the ash content varied over a full year. A seasonal variation was suspected, so classifications were conducted every three months. For each of these, one week was spent on site so that the classification for a season was an average of five consecutive days in that season. Historical data from the weighbridge was also studied so that generation rates could be confirmed. See Table 1 and Figures 1&2.

5.2. RESULTS

The Ratanda data tended to agree with previous work done in the area (Stewart Scott, 1991, Gibbons, 1992) and with work done in China (Rushbrook, 1988). The waste classifications done in India and Nigeria (Rushbrook, 1988, CalRecovery, 1994) however, showed a different composition, with the major components in the refuse being organic putrescible. This difference could be ascribed to climatic differences, fuel type, cultural and possibly even religious differences. A clear seasonal variation in the ash content can be seen, from a peak of 79% of the total in the winter to a low of 54% in mid-summer. A high soil content was noticed throughout due to yard sweeping and the use of front end loaders to collect loosely dumped waste, these tend to dislodge large amounts of soil while collecting the waste. The other components of the waste stream tend to stay stable in terms of quantity, although when expressed as a percentage they increase as the ash content decreases. The Heidelberg data

also agreed with previous work done in the East Rand area as well as with work done in the United Kingdom and in America. Over the full year of classification the composition in Heidelberg did not change significantly. The important biodegradable component, i.e. paper and putrescible material, stayed almost constant at around 65% by wet weight. Generation rates computed from the data were 0.36 kg/person/day for Ratanda and 1.2 kg/person/day for Heidelberg. Loose densities were approximately 400 kg/m³ for Ratanda waste and 120 kg/m³ for Heidelberg.

Table 1 : Summary of refuse compositions and comparisons with other sources.

(a) Developing countries and communities

FERR	NON-FE	GLASS	TEXT	PLAST	PAPER	PUTRES	ASH	MISC	ORIGIN	REF
2.0		2.0		3.0	3.0	2.0	84.0	4.0	EAST RAND	STEWART SCOTT 1991
1.6		2.0		2.4	3.4	4.6	82.0	2.4	EAST RAND	GIBBONS 1992
1.4	0.1	1.3	0.7	2.1	1.4	9.4	78.8	5.0	RATANDA	JUNE 1994 OWN
1.0	0.2	2.0	0.7	3.2	1.9	16.2	69.5	5.1	RATANDA	OCTOBER 1994 OWN
2.1	0.4	1.3	1.3	4.8	5.2	28.8	54.0	2.1	RATANDA	FEBRUARY 1995 OWN
1.3	0.3	1.6	0.5	1.8	2.3	17.4	71.5	3.3	RATANDA	MAY 1995 OWN
1.0		1.0		1.0	6.0	47.0	44.0		INDIA	RUSHBROOK 1988
0.5		0.6	0.6	0.5	2.0	16.0	78.0	1.8	CHINA	RUSHBROOK 1988
2.5		0.6	1.4	4.0	6.6	76.0	8.9		NIGERIA	CALREC. 1994
0.1		0.2	3.0	1.0	2.0	74.7	19.0		INDIA	CALREC. 1994

(b) Developed countries or communities

FERR	NON-FE	GLASS	TEXT	PLAST	PAPER	PUTRES	ASH	MISC	ORIGIN	REF
7.0		10.0		11.0	20.0	29.0	9.0	14.0	EAST RAND	STEWART SCOTT 1991
6.0		4.0		17.0	37.0	32.0	0.0	6.0	EAST RAND	GIBBONS 1992
8.7	0.9	14.7	1.1	11.6	21.1	41.1	0.2	0.7	HEIDELBERG	JUNE 1994 OWN
7.7	2.2	7.2	1.6	13.3	23.7	41.4	0.0	2.9	HEIDELBERG	OCTOBER 1994 OWN
9.1	3.4	2.8	0.7	12.7	28.1	42.8	0.0	9.4	HEIDELBERG	FEBRUARY 1995 OWN
5.7	7.3	5.7	0.4	12.6	21.1	45.4	0.0	1.8	HEIDELBERG	MAY 1995 OWN
6.8	3.2	10.3	1.9	14.4	21.8	40.1	0.0	1.5	SANDTON	AUGUST 1994 OWN
13.0		9.0	4.0	10.0	34.0	22.0	4.0	4.0	USA	RUSHBROOK 1988
8.0		10.0	3.0	7.0	29.0	25.0	18.0		UK	RUSHBROOK 1988

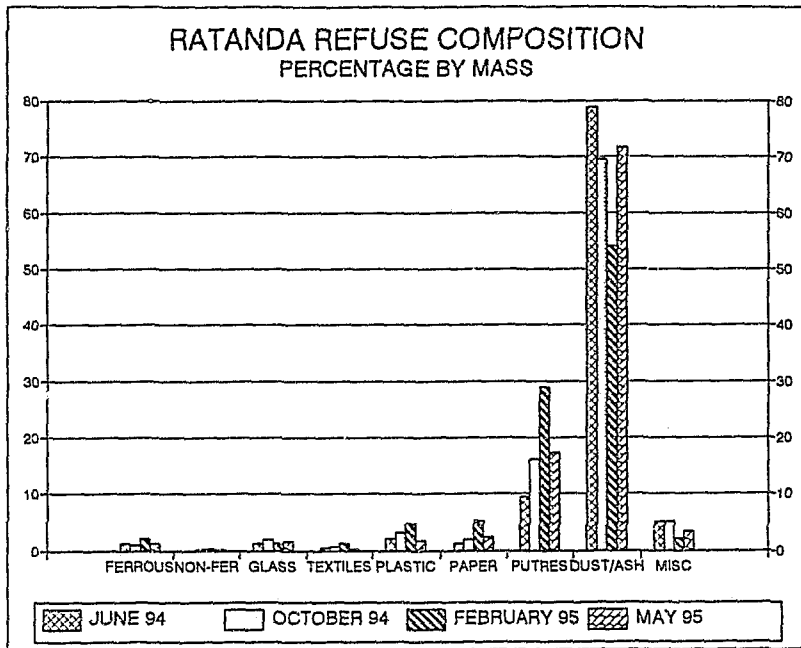


Figure 1 : Yearly variation of Ratanda waste.

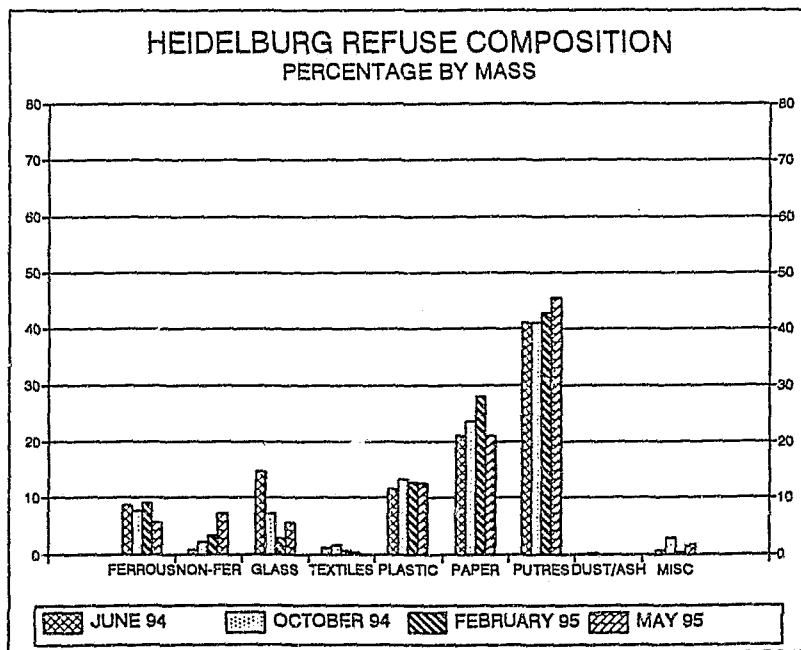


Figure 2 : Yearly variation of Heidelberg waste.

5.3. ASH TESTS

As the ash component is expected to have a large influence on refuse decomposition, a range of tests were performed on this component once it had passed a 4.25 mm sieve. As expected the ash has a high pH which should help to establish rapid methanogenesis. The ash also contains significant amounts of Phosphorous and Nitrogen which are needed as macronutrients by the methanogenic bacteria. Among other trace metals there are very high levels of Iron, Aluminium and Calcium. The effect that these will have is as yet unknown, although it is suspected they could influence alkalinity.

To determine particle sizes of the ash, sieve and hydrometer tests were performed on the ash samples. The ash consisted of over 40% fines (< 0.06 mm)(Craig, 1987). This fines component could have an important influence on the water retaining ability of the waste and hence influence the field capacity. Other tests that still need to be performed are permeability and leaching tests on the ash component. These tests are currently in progress.

6. LYSIMETER STUDIES

Lysimeter studies in the laboratory to model landfill behaviour have become accepted practice for investigation of waste decomposition (Cossu et al, 1987, Rees, 1980, Farquhar and Rovers, 1973, Owens and Chynoweth, 1992). The system has advantages in that the waste is allowed to degrade in a controlled environment with the products of decomposition readily available for analysis. Lysimeter studies will be used to answer the main question in the investigation ie. how does refuse composition affect polluting potential? A typical lysimeter consists of five basic components, namely: a suitable container; an irrigation system; a leachate collection and sampling system; a gas collection and sampling system and a temperature control system.

In order to test the design of lysimeters and the associated backup equipment two prototype lysimeters were constructed and filled with refuse from the town of Sandton in South Africa (See Table 1 for analysis). The refuse was hand sorted and classified in order to find its composition. To negate scale effects, and to stop preferential seepage pathways developing in the waste, the waste was shredded by hand before it was placed in the lysimeter. The shredded waste was then mixed together and lightly compacted into the lysimeters.

Previous lysimeter work done in the Department of Civil Engineering used 220 l HDPE drums with a sealable lid. However, on site and in the lab these were found to be difficult and cumbersome to work with as well as being expensive. The prototypes for this study therefore consisted of one 220 l sealable lid HDPE drum and one 50 l sealable lid HDPE drum. All the other systems on the drums remained the same. The main aim of the study was to see if there was any marked scale effect between the two sizes. If there was no difference then the 50 l drum would be used in the main lysimeter study. The study was also used to see if, with a similar waste composition, similar results to overseas lysimeter studies could be obtained.

The irrigation system consisted of a spiralled hose-pipe that was glued to the inside of the drum's lid. Both the ends of the pipe were sealed and 1 mm holes were drilled at 50 mm

intervals on the underside of the hose-pipe. While the pipe was being glued onto the lid a hole was drilled into the top of the pipe which was connected to a bulkhead fitting. This in turn was connected to a ball valve on the outside of the lid. A modified syringe connected to an inverted 2 l plastic bottle acted as the water delivery system. The syringe was screwed into the closed ball valve and the combination held in place by a modified retort stand resting on the lid. The bottle was filled with the required amount of water and then the valve opened. Before all the water drained out the valve was again closed so that no air was entrained. The system delivered water evenly over the surface at an average flowrate of 20 l/hr. For the purpose of the study distilled water was added to the drums. The water was added at random intervals in the same volumetric ratio to the two drums ie. 0.5 l to the 50 l drum and 2.2 l to the 220 l drum. Leachate was allowed to pond in the leachate collection layer and samples taken at convenient intervals and analyzed.

Leachate collected at the bottom of the drum in the leachate drainage layer, which consisted of a 50 mm deep layer of 22 mm washed andesite stone, and was then drained off by a bulkhead and ball valve placed at the bottom of the drum. More water was added than leachate removed so that the leachate accumulated in the bottom of the drum, its level being shown by the level indicator pipe. This pipe also assisted in drainage of the leachate by breaking any airlocks and equalising the pressure inside and outside the drum. In order to get the leachate out of a sealed environment a pressure differential was created by the use of a nitrogen filled bag attached to the collection bottle.

The gaseous products of decomposition collected in the headspace of the lysimeter. Normally these vented off to the atmosphere through a water filled column which again prevents oxygen from entering the system. When the gas was to be sampled the drums were closed off for approximately one hour. The gas was then extracted and captured in a gas bomb. The extracted gas was analyzed for relative percentages of O_2 , N_2 , CO_2 and CH_4 . A method of determining quantities of gas produced was also developed although only late on in the study. The drums were closed off for a day and the resultant gas pressure buildup was measured by a manometer tube. Barometer readings were taken and the pressure and time used to determine a gas flowrate.

The heating system consisted of three separate components, two external and one internal. The external equipment consists of an electric heater tape which was wrapped around the outside of the drum in a spiral. The drum was then wrapped in 50 mm thick glass wool which acts as insulation. The heater tape was connected to a temperature controller which was set at a specific temperature i.e. 30°C. The controller in turn was connected to an external temperature sensor on the outside of the drum. Feedback between these systems keeps the drum at a constant temperature. Independent of this system was a thermocouple wire placed directly into the waste and emerging on the outside of the drum through a sealed septum in the lid. The thermocouple was connected to a thermocouple bridge so that a separate reading can be taken of the temperature in the waste. These readings were taken daily at the same time. Due to the way an existing lysimeter study was set up and a lack of spare heater tapes the two prototypes were run at slightly different temperatures. The 220 l drum was run at an average temperature of 23°C while the 50 l was run at an average temperature of 28°C.

6.1. GAS DATA

A pattern of events assumed to be typical for gas production in a landfill was presented by Farquhar and Rovers, 1973. Its development was based on the assumption that after placement, aeration of the waste does not occur and that conditions within the decomposing refuse are sufficiently anaerobic to encourage and sustain CH_4 production.

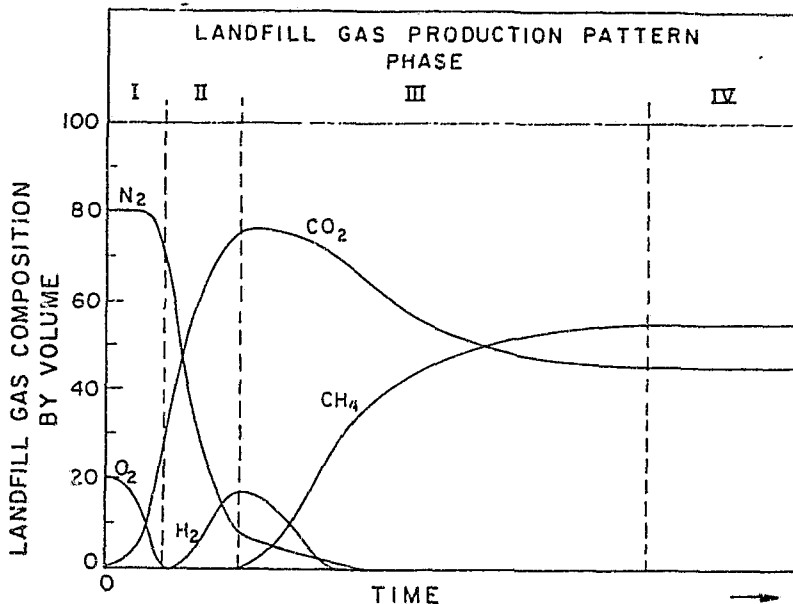


Figure 3 : Landfill gas production patterns.

The four phases were identified as follows :

- Phase 1 - Aerobic
- Phase 2 - Anaerobic non methanogenic
- Phase 3 - Anaerobic methanogenic unsteady
- Phase 4 - Anaerobic methanogenic steady

In phase 1 aerobic decomposition takes place with the consumption of oxygen present in the refuse at placement. CO_2 is produced in approximate molar equivalents to O_2 consumed, with only a slight displacement of N_2 . When all the available O_2 is depleted the gas production pattern enters phase 2 where anaerobic activity becomes dominant, mainly the process of acid fermentation. During this phase a CO_2 peak is produced and H_2 is produced with N_2 being displaced. The decomposition then enters phase 3 where CH_4 production starts and heads to a steady terminal value. H_2 is consumed as it is produced and CO_2 and N_2 values reduce to some terminal value. Phase 4 is stable methanogenesis with 50 to 70% CH_4 and 30 to 50% CO_2 being produced (Farquhar and Rovers, 1973).

Gas data from the analysis of the two lysimeters over a period of 264 days of monitoring are presented below.

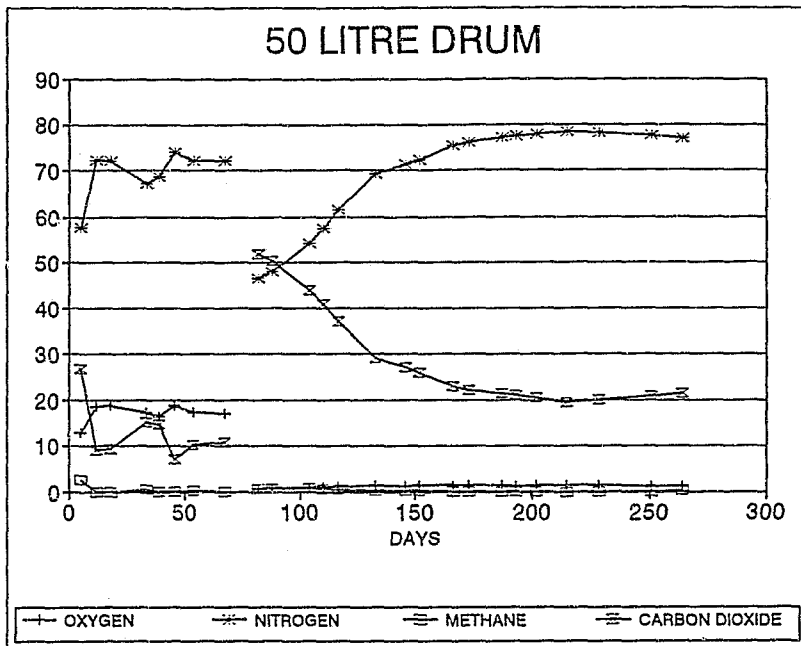


Figure 4 : Gas analysis for 50 l drum.

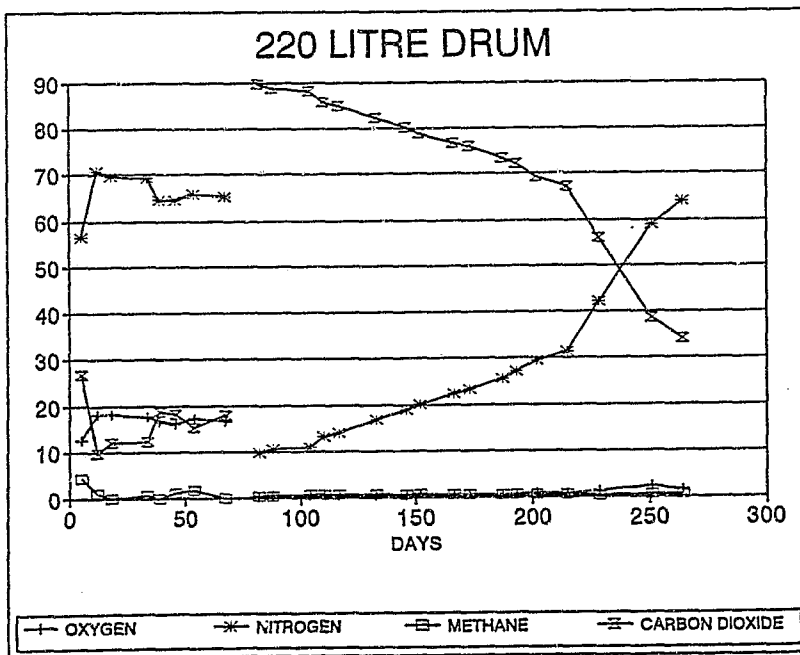


Figure 5 : Gas analysis for 220 l drum.

Due to an incorrect sampling method the data for the first 70 days is not valid. During this period the gas was sampled by evacuating the gas bomb under a vacuum of 5 mm Hg and then opening it when it was connected to the gas sampling port. As a result, when a sample was drawn into the gas syringe it was still under negative pressure and drew in air. This, combined with the fact that the vacuum was not causing a sufficient flow of gas through the bomb for a representative sample to be captured led to the method being changed. A pipette bulb was used to draw gas through the gas bomb at atmospheric pressure. Roughly three times the volume of the gas bomb was drawn through before the bomb was sealed. When the sampling method was first used and the gas analyzed we thought we would have to use the large drums because the gas data was so different for the two sizes. However with longer monitoring we saw that the gas production patterns were the similar, with the time difference being caused mainly by the difference in temperature. What has probably happened in the prototypes is that the CO₂ bloom occurred, ie. phase 2, and then as the environment in the drum was not conducive to methanogenesis the gas composition steadied off to the values shown, with high N₂ and CO₂ and little or no O₂ and CH₄. The low O₂ confirms that the lysimeters are anaerobic.

6.2. LEACHATE DATA

Five samples of leachate were taken at periodic intervals and a complete analysis performed on them. It was decided that more regular tests would not be done due to the cost involved. The results indicated that the pH remained acidic for the duration of the test, whilst COD remained constant at relatively high values. This indicates that although the aerobic and subsequent acid fermentation phase had occurred, the subsequent methanogenic phases had not been established, as can also be seen with the gas data. The reason for this is most likely the placed density of the waste. Rees (1980) found that under conditions of moisture saturation a density of 500 to 1000 kg/m³ was required to establish methanogenesis. The waste in the prototypes was placed at a density of 210 kg/m³. While a high moisture content, temperature and larger surface area (shredding) promotes microbiological activity a high density allows for the controlled hydrolysis and fermentation of refuse polymers to be effectively coupled to CH₄ production. If the placed density is too low hydrolysis and fermentation occur too fast, a very high CO₂ bloom occurs (over 90%) along with the production of simple acids, and the leachate is acidified to such an extent that the environment in the lysimeter becomes toxic to the methane producing bacteria. Another significant trend in the results is the time lag of events between the two drums. Again it is considered that this was due to the temperature difference between the two drums. When the data was compared with acidogenic leachate data from other studies a good correlation was found.

6.3. LYSIMETERS FOR THE EXPERIMENTAL STUDY

Valuable lessons learned in the prototype study will be applied to the full scale lysimeter study. The 50 l drums will be used with only a few modifications from the one used in the prototype study. A separate temperature control system is being installed to allow the lysimeters to run independently of the existing system. When the lysimeters are installed they will be run at 37°C (Farquhar and Rovers, 1973) which is optimal for mesophyllic biological decomposition and is fairly typical of temperatures in landfills. The waste will again be shredded (by an industrial shredder) and when it is placed in the drums it will be

compacted with an electric hammer to densities of approximately 500 to 700 kg/m³. The composition of waste in the lysimeters will be as follows.

Table 2 : Waste composition in lysimeters

Lysimeter No.	Composition	Replicates
1 and 2	Rich waste *	2
3 and 4	Poor waste *	2
5 and 6	50% mixture rich and poor	2
7 and 8	25% rich and 75% poor	2
9 and 10	75% rich and 25% poor	2
11 and 12	2 separate layers rich on top	2
13 and 14	2 separate layers poor on top	2
15 and 16	Poor waste, loose density	2
17 and 18	Poor waste, water deficit	2
	Total No. of lysimeters	18

(*) "Rich" denotes waste from a developed area and "poor" denotes waste from a developing area.

All the mixtures will be by volume with a record of the masses being kept. The "poor" loose density will be run so that a comparison can be drawn with the prototypes which were at "rich" loose density. All the drums will be kept at moisture saturation (except 17 and 18) or field capacity with leachate being withdrawn as it is produced. The leachate will be generated by irrigating with rainwater, which was collected over the rainy season. This was done in an attempt to reduce the number of variables affecting the system. The third world water deficit drums will be run to simulate a landfill in an arid rural setting. The drums will be filled concurrently with the field cells and for significant results should run for at least 200 days. While the drums are being filled the refuse going in will again be classified and samples will be taken for further laboratory analysis eg. water contents, percent volatiles etc. Another significant difference from the prototypes is that cover soil will be placed on top of the refuse before the drums are closed, with a 50 mm pipe to allow for free gas movement into the headspace. As the majority of the methanogenic bacteria are soil-dwelling this should seed the lysimeters with the necessary bacteria.

6.5. MONITORING

The gas analysis on the lysimeters will include the following gases : O₂, N₂, CO₂ and CH₄. Leachate will be analyzed for a shortened list of parameters. These are : pH; conductivity; Eh; COD; alkalinity; ammonia; nitrates; Cl; Ni and K. Both gas and leachate analyses will be performed on a weekly basis. It is also envisaged that mass balances will be performed

on the lysimeters at regular intervals. This will involve disconnecting the lysimeters from their support equipment and weighing them. The variation of field capacity with time is one of the variables that could be found by this means.

7. TEST CELLS

In order to study refuse decomposition on a larger scale a series of test cells is being constructed. They are being constructed with vertical double brick walls and sloping concrete floors, the slope being towards the open side. Wooden fronts will be constructed and treated to close off the open fronts and make two rectangular containers of 3.5 m³ each. Each cell will be lined with geomembrane and have a drainage orifice sealed with a gate valve at the lowest point in order to drain leachate. The leachate drainage layer will be formed by 22 mm washed andesite stone chips in a 100 mm thick layer on the bottom of the cell. The cells will be filled with rich and poor refuse, which will be compacted to a suitable density by a trench compactor. The refuse will be classified before it goes into the cells and samples taken for further laboratory analysis. The waste will then be covered by 200 mm of organic-rich soil which should act as a biofilter and reduce odours. Records will be kept of all water added to the cells as well as any rainfall that falls on the cells. The amount of leachate produced will also be recorded so that the field capacity of the waste can be found. Monitoring of the cells will again include sampling of gas and leachate, with the internal temperature being monitored by thermocouples.

8. FIELD CELLS

In the early stages of the project, field cells were identified as an important component of work to be undertaken. These will be used, along with the test cells, to calibrate the laboratory data with a full scale landfill operation. They provide a means of extrapolating data from the laboratory to the field. Before work on field cells could begin a suitable site was selected near the town of Benoni. The site was chosen mainly due to the separate collection and transport of waste from two predominantly developed and developing communities. Although preliminary studies on waste composition were completed in the Heidelberg area it is felt that the same waste source must be used for the laboratory lysimeters, test cells and field cells. Classification at the Benoni site will be conducted when the cells are filled, possibly coincidental with another Heidelberg classification so that some comparisons can be drawn between the two sources of waste.

Four field cells are in the process of being constructed on the site, one containing mainly rich waste, one poor waste and two with an even mixture of both. To be able to tie the field work in with the laboratory work it was decided to use a 50% to 50% mixture by volume in the mixed cell while keeping note of the mass ratio in the cell as well. A record of the total mass of waste going into each cell will also need to be kept. The final field cells will contain approximately 600 m³ of refuse each.

Leachate generated on the site will be sampled and analyzed as with the lysimeters and test cells. Excess leachate will be disposed of at a sewage works. A water balance will again be conducted to determine the field capacity of the refuse. As part of the operations of the

site there will be a complete weather station which will be monitored on a daily basis to establish water balance data. The station includes a rain gauge, evaporation pan and wind speed and direction recording. A suitable gas collection system will still be decided on, and once it is in place landfill gas will be sampled and analyzed. As the cells are to be kept wet there is a chance that there will be significant gas production. Another parameter to be monitored will be settlement of the waste as it degrades. Two benchmarks will be placed on either side of the cells from where the settlement can be surveyed with a level.

9. LANDFILLING STANDARDS

This section forms the last part of the investigation and is dependant on the results from the previous two components. If it is found that there is a significant difference in pollution potential of waste with respect to its socioeconomic classification then a section on waste stream composition could be included in the minimum requirements. Important factors such as the amount of putrescibles, paper and ash composition of a waste stream could be considered when determining a landfill's classification. This new classification will affect all aspects of a landfill site's development and operation, from siting to closure. It has been suggested that one of the operational aspects that could be changed is the frequency of cover applications (Gibbons et al, 1992). Another factor to consider is the need for compaction of waste with a high ash content.

10. CONCLUSIONS

It is hoped that by undertaking this work that new insight can be gained into the effect waste composition has on the decomposition and stabilisation of landfills as well as quantifying the pollution potential of different wastes. This information will then be used to develop realistic guidelines for the safe and economic landfill disposal of solid waste from developing communities.

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GRADED LANDFILLING STANDARDS TAKING ACCOUNT OF WASTE COMPOSITION

A. B. FOURIE, J. SHAMROCK AND G. E. BLIGHT

*Department of Civil Engineering, University of the Witwatersrand Private Bag 3,
Wits 2050, South Africa.*

SUMMARY: It may be possible to relax standards when low putrescible content waste is landfilled. Experiments using waste from two sources, a low and a middle income community, were conducted using small field cells and laboratory lysimeters. Preliminary results indicate little difference in the quality of leachate from the two waste types. It will be necessary to continue long term monitoring to determine differences in the long term analyses and total contaminant load of the wastes.

1. INTRODUCTION

Although South Africa has an industrialised sector, it cannot be regarded as an industrialised or developed country. The GDP per capita in 1995 was only \$US1800 (South African Institute of Race Relations (SAIRR), 1996). Non-payment for services provided by local councils has become endemic, whilst at the same time the population growth rate in developing communities continues at 2.4%. Spending on service provision has thus to be prioritised carefully. The development of rational and affordable standards for the disposal of Municipal Solid Waste (MSW) by landfilling is one way in which unnecessary expenditure on service provision may be avoided. In terms of South Africa's Environment Conservation Act (Act 73 of 1989), legislative power for the prevention of pollution caused by waste disposal sites is vested in the Minister of Water Affairs and Forestry. To date, control has been achieved by means of a landfill site permitting system. As a further measure, a committee was formed by the Department of Water Affairs and Forestry to formulate a set of Minimum Requirements for the siting, design, operation and closure of landfills. This document has been released as a first draft and is already being implemented by the Department of Water Affairs and Forestry as part of the permitting system to promote compliance with environmental policy and legislation. The objective of the Minimum Requirements is to ensure that the most cost-effective means are used to protect the environment and public health. In this document the requirements for providing an underliner are based primarily on the size and location (defined in terms of climatic variables) of the landfill. Further refinements are envisaged, which take account of the nature of the municipal refuse and how different refuse compositions may translate into different pollution potentials. This latter refinement is the subject of this paper.

2. MINIMUM REQUIREMENTS FOR LANDFILLING IN SOUTH AFRICA

Two criteria are considered when evaluating the need for a MSW leachate collection and management system. These are the climatic water balance index and the size of the landfill.

The climatic index uses published, easily available figures for the weather station closest to the landfill site in question and is defined in terms of a simple atmospheric water balance, with the leachate generating potential being evaluated by the difference between precipitation and evaporation (based on pan evaporation). Factors such as the storage capacity of the waste, runoff and capillary moisture movement are ignored, usually resulting in a conservative estimate of leachate generation potential. A full description is given in Ball et al. (1993) and will not be repeated here.

Cognisance is taken of the fact that small landfills pose a lesser environmental risk than do large landfills, by virtue of the smaller pollution load of the refuse that is stored within them. The Minimum Requirements document distinguishes between four different size categories. It relates the landfill size classification to the maximum predicted rate of refuse deposition for the life of the landfill. These classifications further influence the requirements for installation of leachate management systems at a landfill (Ball et al., 1993).

A third factor that has been identified as potentially influencing the need for a leachate management system is the composition of the refuse deposited at a particular site. It is proposed that waste be classified according to its biodegradable content. If the content of biodegradable material exceeds 20% by dry mass, the waste will be classified as 'W' (high-biodegradable waste), whilst if the biodegradable content is less than or equal to 20% it will be classified as 'w' (low-biodegradable waste). It is envisaged that relaxed standards may be appropriate for 'w' refuse as compared with those required for 'W' refuse. The dividing value of 20% has been chosen relatively arbitrarily and may be modified later. The purpose of this paper is to investigate whether there is sufficient difference between the pollution potential of 'w' refuse and 'W' refuse to warrant the use of different standards for the two waste types.

3. WASTE CHARACTERISATION

It has long been recognised that the composition of a refuse stream depends on the standard of living of the community from which it originates. Table 1 compares waste compositions for refuse from developed (USA and UK) countries with that from developing countries. There are substantial differences in composition, with the richer countries forming 'throw-away' societies in which a large

Table 1. Typical waste compositions (expressed as a percentage of the total), after Rushbrook and Finney (1982) and Blight (1996).

Composition	USA	UK	Delhi, India	Lima, Peru	Kathmandu, Nepal	Wuhan, China	Soweto, South Africa
Vegetable	22	25	47	56	68	16	9
Paper	34	29	6	14	7	2	9
Metals	13	8	1	4	5	1	3
Glass	9	10	1	3	1	1	12
Textiles	4	3	na	4	6	0.5	1
Plastic/leather/rubber	10	7	1	7	1	0.5	3
Wood	4	na	na	na	2	2	-
Dust/ash 'other'	4	18	36	12	10	77	63

proportion of the waste stream comprises of paper, plastic, metals and vegetable matter. In developing countries on the other hand, packaging materials comprise a very small proportion of the waste. In low-income communities that use coal and wood for heating and cooking, the largest component of the waste is usually dust and ash (eg Soweto, South Africa and Wuhan, China), whereas in other low-income communities the predominant component is vegetable matter (eg Delhi, Kathmandu and Lima). In the latter case the waste would also classify as 'w' refuse and be expected to produce high strength leachate. Considering the data in Table 1, it would not seem unreasonable to assume that leachate generated from 'w' waste would differ from leachate generated from 'w' waste, which would be associated with coal and wood burning communities.

To investigate this hypothesis further, it was decided to carry out a characterisation study of two waste streams that could potentially be used in future experimental work. A transfer station was located that accepts waste from both a very low income and coal and wood burning community (C1) and from a medium income community (C2). Additional details are given in Table 2.

Table 2. Details of waste streams used in characterisation study

Waste stream source	Population	Population density (people per hectare)	Refuse generation rate (kg/person/day)	Refuse density (kg/m ²)
C1	35 000	118	0.4	340
C2	15 000	13	1.6	200

3.1 Characterisation methodology

A characterisation of the above two waste streams was done at the point where the waste arrived at the transfer station. Based on the work of Brunner and Waller (1986) and Leroy et al. (1992) it was decided to characterise the waste into nine categories: ferrous metals, non-ferrous metals, glass, textiles, plastics, paper, vegetable and putrescible, dust and ash, and miscellaneous (which included builders' rubble, unburnt coal, leather and rubber). Characterisation studies were done over four one-week periods, one for each season of the year. During each particular week, waste was characterised for five consecutive days (according to the procedure described below) and the results that are presented in the following section are average values for these five day test periods.

The waste from C1, which arrived loose at the transfer station, was collected into two to three 35l buckets, giving a mass of about 20-30kg for each daily sample. The waste was sieved through a 15.8mm sieve into a container for only dust and ash, the remainder being hand sorted into one of the remaining eight categories. A 15.8mm mesh was chosen to speed up the classification process, although it was found that nearly 80% of the material passing this sieve also passed the 4.75mm sieve. The different categories were weighed and the composition determined on a wet weight basis.

The C2 waste arrived in refuse bags in a 19m³ compactor vehicle. Three to four refuse bags (with approximately 85l volume) were classified per vehicle. Only unbroken bags were selected and after breaking the bags open, their contents were sorted into the nine categories listed above.

3.2 Results of characterisation study

The results of this characterisation study are summarised in Figures 1 and 2 for sources C1 and C2 respectively. The low income community, which relies primarily on coal and wood burning for cooking and heating, can be seen to produce a very large quantity of ash and dust, irrespective of the season. Whilst there is some decrease in this component of the waste during summer, it never falls below 50% by wet mass. The putrescible content rarely exceeds 20% by mass of the total and may, according to the definition outlined earlier, be classified as a 'w' composition refuse.

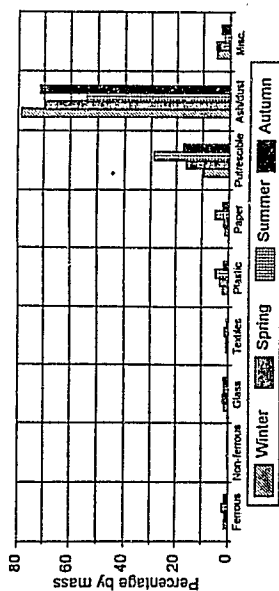


Figure 1. Characteristics of waste from source C1.

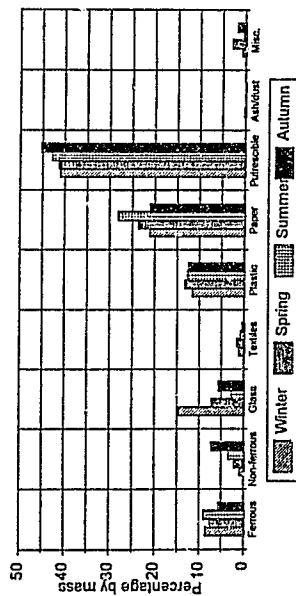


Figure 2. Characteristics of waste from source C2.

The higher income community produces waste that is more similar to that usually found in developed countries. There is always in excess of 60% by mass of paper plus putrescible (ie biodegradable) matter. This waste is thus substantially in excess of the 20% biodegradable matter limit used to define a 'W' waste stream. The two waste streams C1 and C2 are substantially different and thus ideal for evaluating whether different waste compositions will generate vastly different leachates.

4. DETAILS OF EXPERIMENTS

The experimental work was planned to comprise of three components: large field experiments, small field experiments and laboratory lysimeters. The large field experiments are still under construction as a result of delays caused by wet weather and there are no results to report here. When completed, the four field cells will each have a refuse volume of approximately 350m³. One cell will be filled with C1 waste, one with C2 waste and the remaining two cells with a blend of the two wastes.

The small scale field experiments were constructed on the campus of the University and consisted of 3.5m³ containers, 3.12m² in plan area, constructed of vertical double brick walls with

concrete floors sloped towards the front of the container. Each cell was individually lined with a high density polyethylene (HDPE) geomembrane, the base of which was covered with a 100mm thick gravel drain prior to placement of waste within the cell. Each cell had a single base drainage orifice into which the geomembrane was sealed. Waste from sources C1 and C2 was transported from the transfer station and compacted into the cells using a two-person hand-held petrol driven compactor. By weighing the mass of waste placed in each compacted layer, it was possible to control the final density to as close to that in the landfill as possible. After compaction, the waste was covered with a 150mm thick layer of organic topsoil. This soil was not chosen to be representative of a typical capping layer, which would have a much lower permeability, but primarily to serve as an odour barrier. No run-off was allowed from the top surface of these cells (ie all precipitation either infiltrated the cover or was evaporated) and daily measurements of rainfall were made using two rain gauges.

The laboratory lysimeters were made from 500 HDPE drums with sealable lids. They were fitted with a base drainage layer, from which leachate could be extracted via a leachate drainage port. Waste was compacted into the drums using a small hand-held compactor and a small headspace was left above this waste, prior to sealing the drums. Gas samples could be withdrawn from this headspace via the gas sampling port. The lysimeters were vented to atmosphere via a water bath. An irrigation system was also installed in the lid of each lysimeter, through which rainwater was added at regular intervals. A total of twenty lysimeters were commissioned, consisting of two replicates of ten different waste compositions. Only results from lysimeters filled with either C1 refuse only or C2 refuse only are presented in this paper.

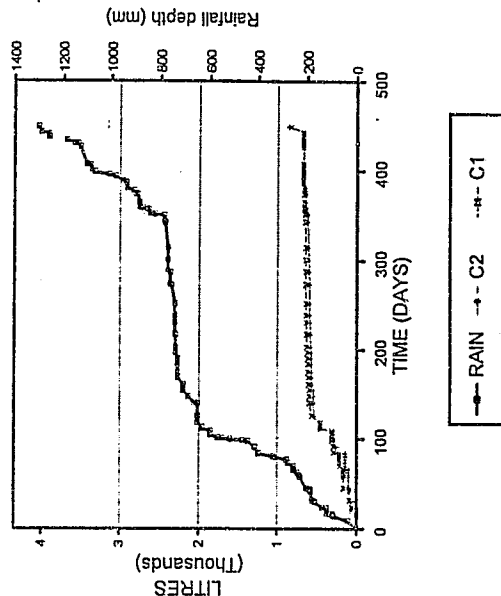


Figure 3. Measured precipitation and generated leachate for small field experiments.

5. EXPERIMENTAL RESULTS

Figure 3 shows the variation with time of the rainfall on the small scale field experiments, expressed in litres (calculated by multiplying measured rainfall depth by surface area of a cell). Also shown is the measured (cumulative) volume of leachate from each cell. There was virtually no difference between the two cells, although waste C1 did begin producing leachate about three weeks earlier than waste C2, which implies that it has a lower field capacity, as would be expected. During the dry winter period (approximately from day 180 to day 340), the leachate production became zero for both cells. Of particular note, however, is the time lag between when rainfall began again (±day 350) and when leachate again began to emerge from the cells (about day 440). This means that a volume of about 1500l (or an equivalent rainfall depth of 480mm) was stored within a cell before leachate again began to emerge from the base of the cell. This means that up to day 340, about 220mm of leachate had been removed from the system, leaving 550mm of the 770mm accumulated rainfall absorbed by the waste. Of this, 480mm (87%) was re-evaporated.

The fact that such a large volume of rainfall was accommodated by the waste before leachate was generated, means that the waste body must have dried out during the dry winter period. This is consistent with findings reported by Rousseev (1995), who found that in the semi-arid climate of Johannesburg, drying of waste could occur to depths of up to 5m. This result lends further credence to the approach adopted in the South African Minimum Requirements document that allow a relaxation of standards for landfills in 'water-deficit' areas, (Ball et al., 1993).

To evaluate the pollution potential of the two waste streams, it was decided to monitor the variation of the following parameters in the leachate emerging from each cell: pH, COD, alkalinity, ammonia,

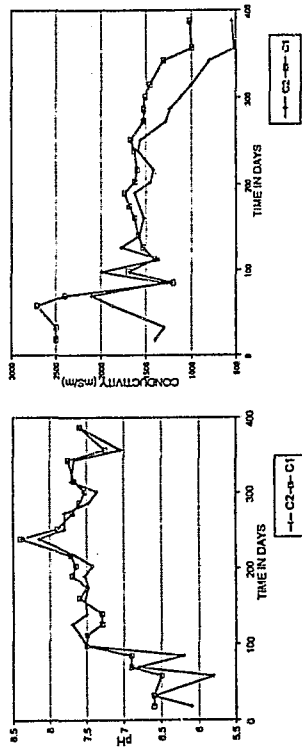


Figure 4. Variation of pH with time.

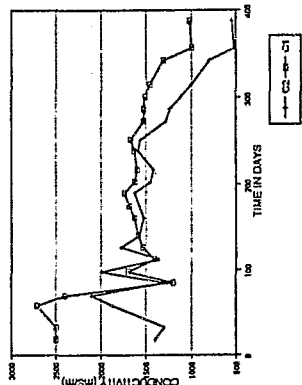


Figure 5. Variation of conductivity with time

nitrate, chloride, conductivity, nickel and potassium. This range of parameters was felt to provide a balance between inorganic and organic indicators, with pH and conductivity being used as crude (and inexpensive) barometer indicators. It was felt necessary to measure at least one heavy metal and nickel was chosen because of its prevalence in previous South African studies of landfill leachate quality (Röhrs, 1997). Figures 4 and 5 show the variation of pH and conductivity with time for the two cells. The time zero corresponds to that in Figure 3. The pH in both cells was initially low,

being less than 6.5 and probably corresponds to the autogenic phase of leachate production. Thereafter the values rise relatively quickly, reaching pH=7.5 after about 100 days and then fluctuating between 7.5 and 8.0. Aside from the first few high values, the conductivity readings remained stable at around 1500µS/m until about day 250, when the values gradually began to decline. This could indicate that the leachate was becoming benign at this time. However, concentrations of chloride and COD (not shown here) showed very marked peaks after about 100 days and then dropped rapidly to equilibrium concentrations at about day 120. These discrepancies serve to demonstrate the complexities of the biochemical processes that take place within a landfill and the need to consider a range of parameters when considering the status of decomposition processes within a landfill.

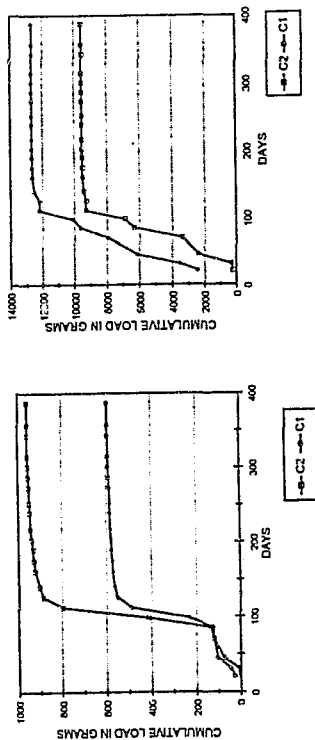


Figure 6. Ammonia load in leachates.

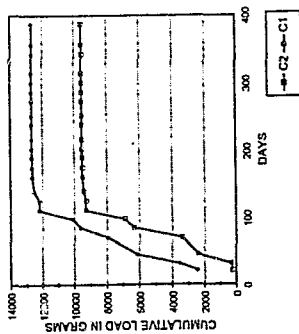


Figure 7. COD load in leachates.

The difference in the quality of leachate produced by the two waste streams has been calculated in terms of the total load of each contaminant (as opposed to a variation in concentration with time). Results are given in Figures 6 and 7 for ammonia and COD. Surprisingly, the low biodegradable content waste produces a higher COD load than waste with a high content of paper and vegetable matter. This difference appears to be relatively constant and manifests itself very early, with the initial leachate samples (at day 40) already showing this marked difference. Even if there had been an error in the initial determination of COD for waste C1, the close similarity between the two sets of data is unexpected. In contrast, the ammonia loads are consistent with what would be expected intuitively. Waste C2 produces an ammonia load that is some 40% greater than for waste C1. The quality of leachate that is produced in the two cells will be monitored for at least another year, with a view to determining whether or not the contaminant load stabilises in one leachate before the other. It is possible that the difference in the ammonia load may increase with time if, for example, the ammonia concentration in the C1 waste begins to decrease as all the contaminants are flushed out.

Figures 8 and 9 show the variation of chloride and potassium load respectively with time for the two cells. Chloride loads are surprisingly similar, whilst the potassium values differ by a factor of two, with the C2 waste having higher loads in both figures. The variation of nickel load is shown in Figure 10 and it is the only contaminant that shows a very substantial difference between the two waste streams.

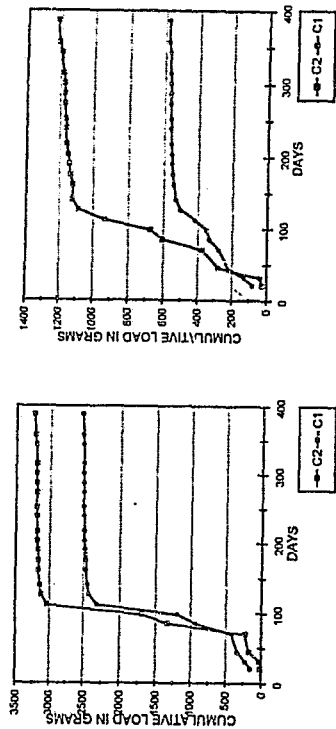


Figure 8. Chloride load in leachates.

Figure 9. Potassium load in leachates.

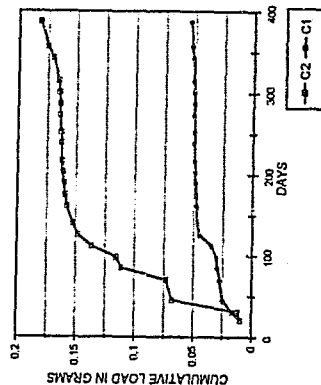


Figure 10. Nickel load in leachates.

6. DISCUSSION

Based on the results shown in Figures 4-10 it is not possible to conclude that 'poor' waste has a lower contamination potential than 'rich' waste. Whilst there is a relatively consistent trend that indicates contaminant loads from 'W' category waste is higher than that from 'w' waste, the difference measured to date is not sufficient to warrant having different standards as regards leachate collection and management systems (and thus different liner systems). However, it is essential that the long-term difference in leachate quality between the two waste streams be determined, because it may happen that the C1 waste becomes stable and benign long before the C2 waste. If this were to be true, then different standards may still be defensible and the above results would thus represent a short-term aberration. Results from the field cells that are currently under construction should also go a long way to answering this question, although the waste will take longer to stabilise because

of its greater volume.

Results from the laboratory lysimeters are, at this time, of limited value. The two lysimeters containing C1 waste have begun to generate leachate, whereas those containing C2 waste have not. This is consistent with the two small field cells, where the field capacity of the C2 waste was greater than that of the C1 waste. This in itself may be a significant finding. C1 waste may reach a benign state much more rapidly than a C2 waste, not only because of an inherently lower contaminant load potential, but also because less moisture is retained per unit volume and thus more leachate is produced than is the case for C2 waste. Waste from a C2 source may thus pose a much longer term environmental risk than C1 waste. This remains speculative.

Analysis of the gas from lysimeters containing C2 waste illustrates a difficulty with lysimeter testing. After 350 days, the gas still contained about 70% carbon dioxide, with the balance comprised of nitrogen. There was no methane present in the gas. The two lysimeters containing C1 waste were less problematical, as a representative density was relatively easily achieved (due to the ash and dust present in the waste stream). After 350 days, the biogas from these lysimeters contains virtually equal percentages of methane and carbon dioxide. Leachate began emerging from these lysimeters after 240 days (whereas after 350 days the lysimeters containing C2 waste had still produced no leachate), and initially had a low pH (average value of 6.2). Contaminant concentrations were also extremely high (eg COD of 120g/l, ammonia of 1800mg/l and chloride values of 3000mg/l). Despite the relatively low biodegradable content of the waste, it appears that the initial pollution potential is similar to that of C2 waste and that leachate manifests earlier than for C2 waste, because of the C1 waste's lower field capacity. Continued monitoring of the lysimeters should indicate whether the contaminant potential of the C1 waste decreases and stabilises quicker than the C2 waste. The results to date, however, confirm that there is not sufficient grounds to adopt different leachate management standards for 'W' waste and 'w' waste.

7. CONCLUSIONS

A characterisation study of waste derived from two economically diverse communities illustrated large differences in waste composition. Waste from a low-income community comprised in excess of 50% by wet mass of ash and dust, with the putrescible content never exceeding 20%, regardless of the season of the year. Waste from a medium-income community usually had in excess of 60% by wet mass of biodegradable matter.

Waste from these two sources was compacted into small field experimental cells and laboratory lysimeters. Monitoring of the leachate from the field cells for the first eighteen months of the study has indicated very little difference in quality. Although the high biodegradable content waste has to date produced leachate with a higher contaminant load than the low biodegradable content waste, the differences are less than a factor of two, except the nickel load, in which case the difference is a factor of nearly four.

Large field cells are currently under construction and it is essential that leachate from these cells, as well as the leachate from the small field cells and laboratory lysimeters continue to be monitored to determine whether or not there will be a difference in the long-term contaminant load in the leachate from the two waste sources.

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