A LABORATORY INVESTIGATION OF THE EFFECTS OF WATER CONTENT AND WASTE COMPOSITION ON LEACHATE AND GAS GENERATION FROM SIMULATED MSW

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

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DECLARATION

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

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(Signature of Candidate)

………….day of……………2012

ABSTRACT

Leachate emitted by landfilled municipal solid waste may cause many and cumulative adverse effects ranging from health problems to environmental impacts. South Africa is one of the few countries in the developing world to have a sound regulatory framework for waste disposal by landfill. Municipal solid waste, however, is not differentiated by content. It is well established that the characteristics of waste produced by affluent suburbs are very different from poor suburbs and informal settlements. The regulatory framework for landfilling is presented as a set of the Minimum Requirements that specifies the design of landfills on the basis of the potential to generate leachate and the type of waste being disposed. The current study interrogates aspects of the Minimum Requirements classification system, namely the classification of waste and climatic water balance. The climatic water balance is used as a guiding tool to assess the potential for a landfill to generate leachate. The end result of this classification is a determination of the proposed site falls under B^+ (water surplus) or B^- (water deficient) class. The Minimum Requirements ensure that all sites which fall under the B^+ class must be equipped with an underliner owing to the potential for leachate generation, while B^- sites do not require any underliner since leachate will be only generated sporadically. However, there is no differentiation on the basis of the content of municipal solid waste from rich and poor suburbs. The present study investigates the generation of leachate from landfills situated on the borderline between B^+ and B^- sites, as well as the degradation of refuse having a range of basic constituents, and representing waste from rich and poor suburbs, as well as a mixture of the two.

Laboratory lysimeters were filled with synthetic waste consisting of varying proportions of paper, putrescible material (grass cuttings) and coal ash (power station fly ash was used to ensure consistency). This was intended mimic the waste coming from "poor" and "rich" suburbs in South Africa. The effect of waste types grading from "poor" to "rich" on leachate quality was investigated. It has been found that a content of 60 % of ash on a dry mass basis, characterizing poor waste, has a neutralizing capacity which results in a better leachate quality than waste with little or no ash, mimicking rich waste. It has been

also established that poor waste has lower leachate generation rates than a 1:1 mix of poor and rich waste as well as rich waste alone. A range of water applications was made bracketing the climatic divide between B^+ and B^-. It was also established that poor waste is characterized by high degradation owing to the high percentage of ash as compared to ash deficient rich waste. It was also noted that different standards for landfills receiving either only poor or only rich waste under the same climatic conditions (B^+ and B^-) in the Minimum Requirements may be advantageous. A relaxation of Minimum Requirements for landfills receiving poor waste could significantly reduce the cost of establishing a landfill under this range of climatic water balance conditions.

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Dedicated

To my late brother

Pandelani Silas Mudau (1970 – 2006)

May his soul rest in eternal peace. Glory be to the Lord.

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Symbol

A……………………………………………………………………..Area
ADF…………………………………………………………….....Acid Detergent Fibre
ADL…………………………………………………………….....Acid Digestible Lignin
B (mm)……………………………………………………………Climatic Water Balance
B^-…………………………………………………………….....Water Deficit
B^+…………………………………………………………….....Water Surplus
BOD………………………………………………………………Biological Oxygen Demand
BMP………………………………………………………………Biochemical Methane Potential
C…………………………………………………………………..Cellulose
Ca^{2+}……………………………………………………………….Calcium
Cd^{2+}……………………………………………………………….Cadmium
Cl^-……………………………………………………………….....Chloride
cm…………………………………………………………………..centimeters
CO_2……………………………………………………………….....Carbon Dioxide
COD………………………………………………………………....Chemical Oxygen Demand
Cr^{3+}……………………………………………………………….....Chromium
DS………………………………………………………………......Dissolved Solids
E…………………………………………………………………..Evaporation
Eh…………………………………………………………………Redox Potentials
ET…………………………………………………………………Evapotranspiration
FC…………………………………………………………………Field Capacity
Fe^{2+}……………………………………………………………….....Iron
G…………………………………………………………………..General Waste

G………………………………………………………………....Groundwater

HW……………………………………………………………....Hazardous Waste

H…………………………………………………………………Hemicellulose

h…………………………………………………………………Low Hazardous Waste

HCO3-……………………………………………………………Hydrogen Carbonate

IRD………………………………………………………………Initial Rate of Deposition

kg……………………………………………………………….. kilogram

L…………………………………………………………………Leachate

ℓ……………………………………………………………….....litre

LOl……………………………………………………………....Ignition Loses

M……………………………………………………………......Mass

m…………………………………………………………….......metre

m^2………………………………………………………………...metre square

m^3………………………………………………………………...Cubic metre

mm………………………………………………………………millimeters

mg/kg………………………………………………………….....milligram per kilogram

mg/l…………………………………………………………….....milligram per litre

mg^{2+}……………………………………………………………...Magnesium

Mn^{2+}……………………………………………………………...Manganese

MRD……………………………………………………………...Maximum Rate of Deposition

M_s………………………………………………………………...Mass of Solids

mS/m……………………………………………………………..millisiemens per meter

mV………………………………………………………………..millivolts

Na^{2+}………………………………………………………………Sodium

NDF………………………………………………………………Neutral Detergent Fibre

NH_4^+………………………………………………………………Ammonium

Ni^{2+}……………………………………………………………………..Nickel

ORP…………………………………………………………………….Oxidation Reduction Potentials

P………………………………………………………………………...Precipitation

ρ…………………………………………………………………………Density

Pb^{2+}…………………………………………………………………....Lead

pH………………………………………………………………………..Hydrogen Potential

P043-……………………………………………………………………Phosphate

R………………………………………………………………………....Rainfall

RO……………………………………………………………………….Run off

$S0_4^{2-}$……………………………………………………………………...Sulfate

SS………………………………………………………………………..Suspended Solids

TO……………………………………………………………………….Total Oxygen

T/ha/yr……………………………………………………………………...Tons/hectare/year

TOC……………………………………………………………………...Total Organic Compound

V………………………………………………………………………….Volume

VFAs……………………………………………………………………..Volatile Fatty Acids

XOCs……………………………………………………………………...Xenobiotic Organic Compounds

Zn^{2+}……………………………………………………………………..Zinc

ΔU_s…………………………………………………………………………Water absorbed and retained by soil cover

ΔU_w…………………………………………………………………………. Water absorbed and retained by refuse

ABBREVIATIONS

APHA……………………………………………..American Public Health Association

ASCE……………………………………………...American Society of Civil Engineers

DWAF……………………………………………..Department of Water Affairs and Forestry

MSW………………………………………………Municipal Solid Waste

MSWM…………………………………………….Municipal Solid Waste Management

SWANA…………………………………………...Solid Waste Association of North America

UK…………………………………………………United Kingdom

USA………………………………………………..United States of America

USEPA……………………………………….......United States Environmental Protection Agency

CHAPTER ONE: INTRODUCTION

1.1 BACKGROUND

Despite the tremendous growth in technical know-how in the field of waste recycling and reuse, the generation of all forms of waste continues to increase worldwide. Municipal Solid Waste (MSW) accounts for a considerable percentage of total waste generation on a global scale (but is a far lesser amount than mine waste). As a result, the field of waste management has struggled to understand some of the overriding factors responsible for the high rate of waste (MSW) generation. Research has shown that an increase in the level of urbanization and population in most parts of the developing countries has played a vital role in escalating the rate of waste generation. Inevitably, this has generated an urgent need for both the responsible authorities and engineers to seek pragmatic approaches to waste management practices. In the last century, dumping was widely used for waste disposal, especially in developing countries. This type of operation involves direct discarding of waste on unused land or into existing holes which in some cases may extend to the water table. Contamination by leachate from the waste is almost inevitable, unless the underlying soil or rock strata is highly impervious (Blight, 2010).

A global turning point was reached in 1959 when the American Society of Civil Engineers (ASCE) defined sanitary landfilling for the first time. Sanitary landfilling is a hygienic controlled way in which MSW is disposed of in defined compacted layers and covered with soil prior to depositing the next layer. With sanitary landfilling the MSW rests on an impervious base layer covered with a drainage layer which is tailor-made to collect leachate exiting from the waste. This layer significantly lessens the contamination of groundwater by leachate. In addition, sanitary landfilling reduces problems such as water ingress from rainfall, odour from waste, breeding flies, scavenging birds and animals etc (Blight, 2010). It is, however, important to note that the disposal of waste by dumping still persists in most parts of the developing countries, especially in informal settlements or squatter camps.

This can partly be attributed to the lack of flexibility in regulations controlling waste disposal adopted by third world countries from the first world countries and/or by the absence of any form of requirements for landfilling in developing countries. Thus, Blight (1999) concluded that "Best for the first world may not be best for the third world or even best at all".

Experience in dealing with municipal solid waste storage facilities indicates that poor siting, design and operation have adverse impacts on environment, safety and health. Contamination of surface and groundwater by leachate as well as by wind-blown litter are the most adverse environmental impacts resulting from unsafe practices for waste disposal by landfilling. Water forms an integral part of most living beings and without water there would be no life in existence. Owing to the scarcity of this precious commodity, it is imperative to safeguard water resources from any form of contaminants; whether the contaminants are attributable to either natural or anthropogenic activities. South Africa falls among those countries in Africa that rely on groundwater to a large extent. Therefore, it is important to put proper preventive measures in place to ensure sustainability by securing the groundwater from contamination. Leachate can be defined as water that has percolated through the refuse, and acts as a means for transport of contaminants to pollute surface and groundwater. The production of methane gas from landfills may have an adverse influence on global warming, and is hazardous as it is both flammable and explosive.

Owing to the environmental consciousness of the South African Government, the Department of Water Affairs and Forestry formed a committee in 1990 to draft standardized regulations for the disposal of waste by landfilling. The ultimate goal of this initiative was to formulate a set of Minimum Requirements for siting, design, operation and closure of landfills. The Minimum Requirements take into account the type of waste to be deposited, the projected final size of a landfill and the site climatic conditions prior to siting of any landfill (Blight, 2006). Summaries of two editions (1994 and 1998) of Minimum Requirements for waste disposal by landfilling are presented in the literature review (section 2.3).

1.2 PROBLEM STATEMENT

The South African Government through the Department of Water Affairs and Forestry, is to be congratulated for introducing the unique graded Minimum Requirements which are being adopted by other countries, especially developing African countries, presently without any form of regulatory framework for waste management. This is because the graded minimum requirements take into account the type of waste to be deposited, the projected final size of a landfill and the site climatic conditions prior to siting of any landfill, unlike the rigid regulatory requirements in many developed countries.

The Minimum Requirements allow the site climatic conditions to be determined through the climatic water balance by using annual rainfall and annual A-pan evaporation data available from the weather station nearest the site proposed for landfilling. As stated in section 2.3.2 (c), for all the sites which fall into the B^+ (water surplus) class, it is a pre-requisite to provide an underliner owing to the potential for leachate generation. Conversely, all B^- (water deficient) sites, disposing only general MSW, do not require any form of underliner since the Minimum Requirements assumes leachate will be only sporadically generated.

To date, the Department of Water Affairs and Forestry has published two editions of Minimum Requirements (1994 and 1998), but the issue of rising costs to meet the standards for waste disposal by landfilling remains an issue of great concern to the municipalities (Bredenhann, 2005). So far, there has been no attempt to gain knowledge on landfill sites classified on the boundary between B^+ and B^-, but instead most of these landfills have been classified as B^+ sites. Hence to date South African Minimum Requirements for landfilling do not cater for those landfills situated between B^+ and B^- climatic zones. They are presently regarded as B^+ sites. This could be regarded as one of the contributing factors which is adding unnecessary costs for constructing and operating a landfill, and hence delaying the development of compliant sites. However, a draft of national standard for waste disposal to landfill has been issued for comment suggesting that all the landfills must underlined, either B^+ or B^- landfills (National Environmental Management: Waste Act (59/2008) General notice 432, Government Gazette No 44414 0f 1 July 2011). The present study was conducted in light of two editions of Minimum

Requirements for waste disposal by landfill (1994 and 1998) which is legitimately enforced presently.

1.3 CONTRIBUTION OF THIS WORK

The present study intends to examine the decompositional behaviour of municipal solid waste (MSW) of differing compositions as the effects of climatic conditions are varied from B^+ to B^-; to see to what extent changing simulated climatic conditions and waste composition affect the production and quality of leachate, as well as the decomposition of the waste. Hopefully, the results of this research will lead to the climatic classification in the Minimum Requirements being re-assessed and modified or improved if found necessary. As a result, this may help the local authorities and waste management companies currently operating marginally B^+ landfills to alleviate their financial burden and improve the safety of the environment.

1.4 OBJECTIVES OF THE STUDY

The present study sets out to investigate the leachate generation potential from landfills situated at two different climatic zones, namely B^+ (water surplus) and B^- (water deficient), as well as the decomposition of refuse (MSW) having different compositions under these conditions.

The specific objectives are to:

- Investigate the effects of varying water content and waste composition on gas and leachate production. The range of water contents will bracket the estimated water content for a climatic water balance on the division between B^+ and B^- climatic zones, with one specimen of each composition wet of the B^+ to B^- and one drier. Waste composition will grade between those of ''poor'' and ''rich'' waste in terms of ash, paper and putrescible contents.

1.5 HYPOTHESIS

Varying moisture contents between B^+ (water surplus) and B^- (water deficit) landfill sites will have an influence on landfill emissions as well as the degradation of MSW. Owing to the differences in waste composition of MSW received by a particular landfill; from both ‘‘poor’’ and ‘‘rich’’ suburbs, the poor and rich waste may also exhibit differences in the landfill processes of leachate generation, gas production and waste decomposition.

CHAPTER TWO: LITERATURE REVIEW

2.1 THE PURPOSE OF THIS CHAPTER

This chapter is taken as the basis for the present study since it allows one to gain further knowledge related to the topics of investigation. The industrial revolution in various parts of the world has impacted on the economic growth of many countries drastically, yet the majority of these countries remain poor. As a result, practitioners in the field of waste management have struggled to understand the overriding factors affecting generation of waste (MSW), both in developing and developed countries. It is the purpose of this chapter to review literature on critical issues of waste management in developing countries, with specific reference to refuse generation, collection, disposal and characterization, as well as the adverse impacts posed by unsafe disposal of MSW. The regulatory framework for waste management in South Africa, a developing country, will also be presented since it sets a benchmark for waste management practice in other developing countries. South Africa is one of the few countries in the developing world to have a sound regulatory framework for waste disposal by landfill. The regulatory framework is presented as a set of Minimum Requirements. The Minimum requirements are graded landfill requirements based on a classification system by waste type, landfill size and climate. The present study probes the effects of the first and last of these. The gap in knowledge had revealed that the Minimum Requirements for waste disposal by landfill do not cater for those landfills situated on the borderline between B^+ and B^- classifications, except that they are assumed to have a B^+ classification. Therefore, this chapter presents the regulatory framework for waste disposal by landfill, especially the classification system. Apart from the legislative requirements, this chapter will also review the concept of sanitary landfilling and bioreactor landfills, with emphasis given to sanitary landfill operations, factors affecting leachate generation and the effects of waste composition in a sanitary landfill. Furthermore, factors affecting the decomposition of refuse (MSW) together with the landfill gas production will be reviewed.

2.2 MUNICIPAL SOLID WASTE MANAGEMENT IN DEVELOPING COUNTRIES

2.2.1 Definitions

(a) Developing Countries

To date, there is no internationally acceptable definition of a developing country; the definition may vary from one individual to another or from one scholar to another. According to Torado (1977), a developing country is characterized by the following:

- Low standards of living;
- Low levels of productivity;
- High rates of population growth and dependency burdens;
- High and rising unemployment and underemployment;
- Significant dependence on agricultural production and primary product exports;
- Dominance or dependence and vulnerability in international relations.

In contrast, a developed country is one wherein the aforementioned six points are the opposite. Despite the characteristics defining a developing country as outlined by Torado (1977), an explicit definition which distinguishes a developing country from a developed one remains undecided. It was early in the 1990s when a formal and more succinct definition of a developing country was given by Campbell (1993). According to Campbell (1993), a developing country is one where the per capita gross domestic product is lower than the average for the world as a whole. In simple terms, a developing country is one where the people are poor, on average. However, Blight (1996) argued that this definition may be inadequate to be applied in many countries. The prime reason behind his argument was that the industrialized urban areas in a developing country may be fully developed, whereas country areas are still developing, as for example, in China or India. In addition, it is seldom that wealth is evenly distributed between urban and rural areas. As a result, it is better to refer to these countries as having mixed economies.

(b) Municipal Solid Waste (MSW)

The term municipal solid (MSW) refers to all the waste generated, collected, transported and ultimately disposed of within the administrative boundary of a municipal authority. MSW is composed of household refuse, institutional wastes, street sweepings, commercial wastes, as well as construction and demolition wastes (Although the last two are increasingly being recycled to produce new construction materials).

(c) Municipal Solid Waste Management (MSWM)

Municipal Solid Waste Management (MSWM) encompasses the aspects of waste generation, waste composition, collection, recycling, treatment, and disposal. These management functions are collectively embedded in a site-specific socioeconomic, behavioral, cultural, institutional, and political framework.

2.2.2 Current Situation of MSWM in Developing Countries

The science and engineering of MSWM is a relatively new concept in developing countries. To date, the problems of attempting to bring the municipal solid waste problem under control in developing countries appear almost insuperable, yet we must work towards more efficient collection and transportation and safer and environmentally acceptable disposal of MSW (Blight and Mbande 1996). In developing countries, little attention has been paid to help addressing issues of concern in regard to MSWM, both by policy makers and academics (Medina, 1997).

According to the Global Waste Management Market Report (2007), approximately 2.02 billion tons of MSW were generated globally in 2006, representing a 7 % increment since 2003. The World Bank estimates revealed that municipalities in developing countries spend 20-50 percent of their available budget on solid waste management. These high expenses represent a large expenditure for collecting, transporting and disposal of MSW in Third World countries (Medina, 1997).

Despite the maximum efforts being made by governments and other entities geared towards solving solid waste management-related problems in developing countries; there are still identified major gaps in knowledge to be filled in this discipline

(UNEP, 2009). Hence, research is in progress to seek pragmatic approaches to waste management practices in developing countries.

As has been stated in chapter 1 (section 1.1), the slow improvement of waste management practices in developing countries is partly attributable to the lack of flexibility in regulatory frameworks for waste disposal adopted by third world countries compared to first world countries and/or the absence of any forms of requirements for waste management in developing countries. In this section, the term developing country is used interchangeably with third world country.

According to Blight and Mbande (1996), it is far more difficult, but still possible to solve problems of upgrading practices for the disposal of MSW in developing countries and developed parts of countries with mixed economies than in developed countries and developed areas. There are several reasons for these difficulties, but they all basically arise from poverty and lack of education and opportunity and, in some cases, to the adherence to cultural customs that do not practically or easily fit the modern world.

Despite the large sums spent by municipalities in developing countries to collect, transport and dispose of MSW, only 50-80 percent of the refuse (MSW) generated is collected in third world cities. For example, about 50 percent of the refuse generated in India is collected, 38 percent in Karachi, 40 percent in Yangoon, and 50 percent in Cairo. Apart from refuse collection, waste disposal in major Asian cities has also received less attention, with 90 percent of waste generated ending up in open dumps (Medina, 1997). Recently, it is more evident that the biggest challenge faced by developing countries is the lack of proper collection and disposal of MSW. Therefore, in this section, refuse collection and disposal in developing countries will be reviewed.

(a) Refuse Collection

The conventional collection of refuse is easier in formally laid out towns and cities either using specially designed refuse collection vehicles or general purpose trucks or tractor-trailer combinations (Blight and Mbande, 1996). The informal settlements, squatter camps, or shanty towns that exist within and on the outskirts of the most towns and cities in developing countries are characterized by many refuse collection-related problems. The problems and situation in these communities are significantly different from those in the formal suburbs of developed countries with well-planned roads and a regular layout. A study by Blight and Mbande (1996) pointed out two possible reasons which account for these differences. Firstly, the MSW problems of developing countries are often compounded by lack of awareness of the possible dangers presented by accumulations of refuse between the informal houses. Secondly, there is the lack of access roads within informal settlements which are typically characterized by a random layout of informal houses, with narrow and irregular spaces separating them. As a result, this prevents the usual vehicles from gaining access to provide a conventional house to house refuse collection service. In the same study, Blight and Mbande (1996) also suggested one way to overcome the problems of inaccessibility in informal settlements. This includes provision of large communal containers or skips at strategic positions. However, even this measure offers a range of problems:

1) Refuse may be dumped next to the containers and not within because of the container's height which often prevents children from reaching over the sides, or because they are not emptied regularly.
2) Refuse is not taken to the containers, but rather disposed, for example, by throwing it over the fence, and accumulates in odd corners.
3) Refuse-filled containers are set alight.
4) Moveable or light-weight containers are overturned to provide shelter for homeless people.

It appears that the ultimate solution to solve these problems is to devolve the responsibility for refuse collection to community members, in the form of self-help schemes. Due to high financial constraints, such schemes will not be self-sustaining unless the money is sourced from the taxes of the greater community. In addition, there is potential political opposition to such schemes especially by the wealthier taxpayers and rate-payers (Blight and Mbande, 1996). However, such schemes are in place and successfully used in some parts of South Africa.

A recent study on commercializing solid waste management in South Africa shows that the situation of waste management has improved since 1994. However, door to door refuse collection and teams of street sweepers are still confined to previously whites-only suburbs rather than predominantly black township and rural residents. Inevitably, such residents are compelled to either dump their garbage in open dumps or unsealed communal skips.

(b) Dangers of Unsatisfactory Refuse Disposal

The unsanitary disposal of MSW generates various environmental, safety and health-related hazards. In most parts of developing countries, dumping is a highly utilized and well-tolerated method for disposal of MSW. According to Medina (1997), the lack of refuse collection especially in low-income communities had led to dumping of waste at the nearest vacant lot, public space, creek, rivers, or it is burned in their backyards.

Heavy rainfall and prevailing windy conditions often facilitate refuse the transport of refuse from one locality to another. During rainy seasons, uncontrolled waste accumulated on the streets may clog drains, which may result in flooding. Wastes can also be carried away by runoff water to rivers, lakes and seas, affecting those ecosystems. Alternatively, wastes may end up deposited in open dumps-legal and illegal-, which is the most common MSW disposal method in many third world countries (Medina, 1997).

It is stated above that unsanitary disposal or open dumping generates various environmental, safety and health-related hazards. MSW can adversely affect the health, quality of life and safety of human beings to a greater extent than does mine waste (Blight, 2010). Therefore, the adverse impacts posed by MSW dumping should not be underestimated.

The degradation of the organic fraction of refuse favors the production of methane, which is a highly flammable gas and can cause fires and explosions. In addition, methane and carbon dioxide are major greenhouse gases which, emitted from open dumps, contribute to global warming. In developing countries, scavengers are prone to drowning in carbon dioxide if they fall asleep in a hollow on a landfill, possibly kept clear of gas by the wind during the day, which then fills with gas when the wind drops at night. Cases of this nature have been also reported on coal waste dumps and numbers of people die in this way each year (Blight, 2010).

A case study is documented in South Africa where methane escaped from a MSW dump, passed along the line of a badly backfilled trench, and into the basement of a nearby building, where it accumulated. The electrical spark caused by switching on the light in the basement set off an explosion that wrecked the building (Blight, 2010).

Fires are also safety hazards at open dumps, often arising due to spontaneous ignition of methane by heat generated during biological degradation. In some cases, dump managers intentionally set fires as a means of reducing the volume of waste, in order to extend the life span of the dumps. Human scavengers may also deliberately cause fires in order for them to more easily spot and recover metals among the ashes than among piles of mixed wastes (Medina, 1997).

To date, flow failures of six large-scale MSW dumps and landfills have been documented in the technical literature, between 1977 and 2005 (Blight, 2008). Unfortunately, many of the people living in the vicinity of the dumps end up losing their lives, while others are made homeless. Great damage to public and private property can be caused by flow failures of MSW. A failure of the Payatas waste dump at Quezon city, Philippines in 2000, resulted in burial of houses and informal shack houses located at the toe of the dump. It has been reported that only fifty-eight people were rescued, and after intensive searching and digging, 278 bodies were recovered, leaving between 80 and 350 missing, believed dead (Blight, 2008). Thus, MSW may continuously threaten the health, quality of life and safety of population if not properly managed.

The complex biological and chemical processes that occur in open dumps generate poor quality leachates, which pollute surface and groundwater. In developing countries, inadequate disposal of refuse can be a major factor in promoting the spread of insect-borne and parasitic diseases such as gastroenteritis and malaria (Blight and Mbande, 1996).

In conclusion, Blight (2010) stated that "uncompacted, un-covered dumps of MSW should be landfills of yesterday. Unfortunately, they are still with us today and in developing countries they are too often the only form of MSW disposal in use. Even though it is 50 years since the concept of the compacted, covered sanitary landfill was introduced and formalized, it remains the landfill of the distant future over much of the Earth". Furthermore, Blight (1996) encouraged the application of graded standards for landfills in developing countries in order to improve waste management practices (see section 2.3 in this chapter on regulatory framework for waste disposal by landfill in South Africa, a developing country).

2.2.3 Refuse Generation and Composition in Developing Countries

(a) Refuse Generation

Research shows that refuse generation per head of population varies from one country to another, with varying rates of refuse generation in different parts of a country. Several studies have been conducted to investigate the factors affecting refuse generation and composition, both in developed and developing countries (e.g. Mbande 2003; Blight and Mbande 1995, 1996; Medina 1997; Rushbrook and Finnecy 1988; Shamrock 1998).

A literature review shows that wealthier communities usually consume more than lower-income communities, which results in higher waste generation rates for wealthier communities. Wealthier communities form 'throw away' societies, while poor 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). This is in substantial agreement with a chapter in a recent text book by Blight (2010), which shows the relationship between the wealth and the amount of waste generated. A study by Medina (1997), also found that there is a positive correlation which exists between a community's income and the amount of MSW generated.

According to Blight (2010), the ever-expanding populations around the world seek a better standard of living that, in turn, demands the acquisition of new goods and discarding of old. In addition, as most of the world's population is urbanized, an ever-increasing volume of MSW is generated. For these reasons, rapid urbanization and population growth have played a significant role in the generation of huge volumes of MSW, especially in developing countries.

A lot of work has been done in order to understand the amount of waste generated in various facets for both developing and developed countries. Among several studies conducted, studies by Blight and Mbande (1996) revealed that the amount of waste generated per head of population varies from as low as 0.05kg/person/day for a poor community to as much as 2kg/p/d for a corresponding rich community. The demographic information of the country and the amount of waste generated per person per day is important when calculating the annual global generation of household waste (Blight, 2010). For instance, if the assumption is made that the world's population is 7000 million and most of these people are poor and on average generate 2 kg/p/d, the annual global generation of MSW would be $7\times10^9\times0.2\times365$kg/y = 511×10^9kg/y.

(b) Refuse Composition

The type and quantity of waste generated by a community depends on its culture and the per capita income (Blight and Mbande 1996). Wealthy communities form throw away societies, while poor communities have minimal goods to throw away and are more ingenious in reusing, recycling and refurbishing articles that a wealthier community would discard.

These circumstances result in significant differences between the wastes generated in communities with different economic circumstances as well as in developing communities in different parts of the world. The differences in waste compositions in developing countries are attributed to the particular nature of the culture, climate, differences in fuel used and dietary pattern. For instance, the majority of Indian civilians are vegetarians and dung is a common cooking fuel. In Wattville (which is a poor suburb in South Africa) coal is commonly used for both cooking and heating, and every combustible material tends to burned as fuel (Blight, 2010). In each case, these customs result in a different composition water content for MSW. Climate is also one of the important factors which contribute to differences in refuse composition from different communities both in developed and developing countries.

A study by Shamrock (1998) revealed the seasonal variation of refuse composition from poor and rich communities in South Africa. The comparative study encompassed two poor communities, namely Ratanda and Wattville, as well as Heidelberg and Benoni, which are rich communities. During summer, the waste compositions of the two poor communities were characterized by a high ash content, forming 60 % of the total waste stream, while rich communities were characterized by 0 % of ash. A significant difference in waste composition from poor communities was noticed during winter with an increase of 15 % of ash fraction, whereas waste composition in rich communities remained at zero percentage.

2.3 MINIMUM REQUIREMENTS FOR WASTE DISPOSAL BY LANDFILL IN SOUTH AFRICA

2.3.1 Historical background of the Minimum Requirements

Historically, the concept of sanitary landfilling originated from the northern hemisphere in the wetter parts of Europe and the United States (Blight and Fourie, 1999). As a result of the wetter climatic conditions in these countries, it has become an expectation that all landfills will generate leachate; therefore, all the landfills are required to be equipped with underliners in order to protect ground and surface water from contamination. However, the regulations in these countries were found to be not applicable and affordable in most poor developing countries. Therefore, there was an urgent need to develop standards suitable for poor developing countries.

The South African Government Department of Water Affairs and Forestry (DWAF) formed a committee in 1990 to formulate a set of Minimum Requirements for siting, design, operation and closure of landfills (Blight, 2006). The set of Minimum Requirements were drawn up for South Africa a developing country mainly comprised of poor communities which cannot afford to pay for the elaborate and stringent standards prescribed in developed countries (Ball et al., 1993).

As a result, some of South Africa's neighboring countries in the South African Development Community (SADC) region, including Botswana, Swaziland and Namibia have developed their waste management and disposal systems in the line with the South African Minimum Requirements for waste disposal by landfill (Ball et al., 1995). The main objective of the graded minimum requirements is to ensure that the most cost effective means are used to protect the environment and the public health from both short and long term adverse impacts of solid waste disposal (Ball et al., 1993; Blight, 2006).

The specific objectives are to:

- avoid degradation of the general environment in which the landfill is situated;
- prevent pollution of the adjacent surface and groundwater regimes; and ensure that the landfilling process is in itself environmentally and aesthetically acceptable.

The objectives set by the Minimum Requirements can be readily achieved by applying a set of graded minimum requirements to different conditions, in a scientifically defensible way. The Minimum Requirements are centered on a landfill classification system whereby a landfill is classified according to the type of waste that the landfill receives, the final projected size of the landfill and the climate (Blight, 2006). The latter is of paramount importance to the Minimum Requirements, and as a result, graded minimum requirements can be applied in other developing countries which have a range of climatic conditions.

2.3.2 Landfill Classification System

As stated earlier, a landfill in the context of the Minimum Requirements can be classified according to the type of waste, size of the landfill and the climate. Table 2.1 below summarizes the landfill classification system.

WASTE CLASS	G General Waste								H Hazardous Waste	
SIZE OF LANDFILL OPERATION	C Communal Landfill		S Small Landfill		M Medium Landfill		L Large Landfill		H:h Hazard Rating 3&4	H:H Hazard Rating 1-4
SITE WATER BALANCE	B^-	B^+	B^-	B^+	B^-	B^+	B^-	B^+		
MINIMUM REQUIREMENTS										

<u>NOTES:</u>

B^- = No significant leachate will be generated in terms of the Site Water Balance (Climatic Water Balance calculations plus Site Specific Factors), so that a leachate management system is not required.

B^+ = Significant leachate will be generated in terms of the Site Water Balance (Climatic Water Balance calculation and Site Specific Factors), so that a leachate management system is required.

h = A containment landfill which accepts **H**azardous waste with Hazard Ratings 3 and 4.

H = A containment landfill which accepts all **H**azardous waste, i.e. with Hazard Ratings 1, 2, 3 and 4.

Table 2.1: A summary of Landfill Classification System (DWAF, 1998).

According to the document of Minimum Requirements for waste disposal by landfill which has been published by the Department of Water Affairs and Forestry (DWAF, 1998), the specific objectives of the landfill classification system are to:

- consider waste disposal situations and needs in terms of combinations of waste type, size of waste stream and potential for significant leachate generation.

- develop landfill classes which reflect the spectrum of waste disposal needs.

- use the landfill classes as the basis for setting graded minimum requirements for the cost-effective selection, investigation, design, operation and closure of landfills.

a). Waste Type

The Minimum Requirements recognize two types of waste, namely General (G) and Hazardous (HW) waste. By definition, General waste is essentially MSW disposed of within the domain of local authorities and mainly comprises of household waste, builders' rubble, garden waste, commercial waste and some dry industrial waste (Rohrs, 2002). General waste may also be characterized by small constituents of hazardous waste such as batteries, insecticides, weed killers and medical waste etc. On the other hand, hazardous waste has potentially severe adverse impacts on public health and the environment, even in low concentrations (DWAF, 1993). This can be explained in terms of the inherent nature of hazardous waste; the chemical and physical properties such as toxicity, ignitability, corrosivity, carcinogenicity etc. Furthermore, hazardous waste can be sub-classified as low hazardous (h) or high hazardous (H).

According to Blight (2006), all the hazardous waste must be disposed of at hazardous waste (HW) landfills, and it is a minimum requirement that such landfills must be equipped with an underliner and leachate collection system. However, landfills receiving general waste (G landfills) pose less demand for installation of an underliner than do hazardous landfills.

(b) Landfill size

The Minimum Requirements classification system distinguishes four different landfill size categories which are: communal, small, medium and large (Table 2.2). According to Ball et al. 1993, the size of the waste storage facility can be assessed based on the three following inter-related sub-criteria:

1. The size of the community served by the landfill;
2. The rate of deposition of waste; and
3. The size of the landfill (either planned or existing)

Landfill Size Class		Maximum Rate of Deposition (MRD) (Tonnes per day)	
Communal	C		<25
Small	S	>25	<150
Medium	M	>150	<500
Large	L	>500	

Table 2.2: Landfill size classes (DWAF, 1998).

Blight (1994) noted that the rate of deposition of waste is the major factor dictating the landfill operation in terms of the need for facilities, plant and operating skills. The size of the landfill is calculated from the maximum rate of deposition (MRD) of waste expected at the landfill site. The maximum rate of deposition (MRD) gives an indication of the total amount of waste expected during the lifespan of the landfill. The following formula is used to calculate the MRD:

$$MRD = IRD\,(1+D)^{t}$$

Where

MRD = maximum rate of deposition after t years (tonne/day)

IRD = initial rate of deposition (tonne/day)

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

t = years since deposition started

It is noteworthy that landfills for general wastes (G landfills) are divided into four size categories once the MRD has been calculated. In case of hazardous waste landfills (H landfills), the size of the landfill is not taken into consideration and their classification is based on the Hazard Rating of waste.

c). Climate

In this section, the background of the water balance method which has lead to the establishment of the climatic water balance method in the Minimum Requirements is reviewed, followed by the presentation of the climatic water balance method.

According to Blight et al. (1995) the potential for a landfill to generate leachate regularly or not can be investigated by means of the water balance method. Literature survey indicates that the use of the water balance method for predicting leachate generation from landfill sites date back to mid 1970s. This involves an outstanding work which had been carried out by Fenn et al., 1975 to predict leachate generation in solid waste disposal sites. Since the proposal of this method, research has been widely conducted to improve or modify the recently known water balance method. A study by Farquhar (1989) has simplified the various components of water balance as depicted in the figure below.

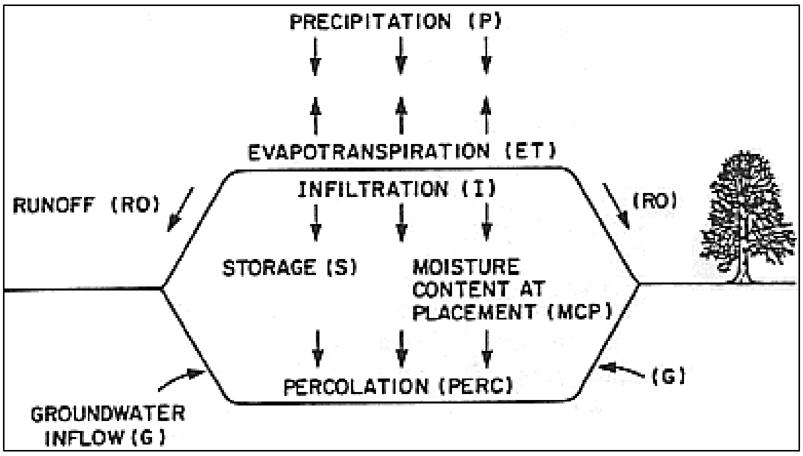


Figure 2.1: A generalized pattern of leachate formation.

Figure 2.1 explicitly depicts a general overview of some of the components of a water balance. Close inspection of this figure indicates that not all the precipitation falling onto landfill surface ends up as an infiltrate, but some of it becomes runoff. Owing to the seasonal variations in different geographic areas, a wide range of temperature prevails. As a result, a certain percentage of infiltration is lost through the process of evaporation. For closed landfills capped with vegetation, some precipitation will be lost through evapotranspiration. In view of this, vegetation has the capability to intercept rainfall prior to reaching the ground surface and re-direct it to the atmosphere as water vapour. For the remaining infiltration to reach the base of the landfill after evapotranspiration (ET) had taken place, there is one major condition to be met before any downward percolation of seepage can take place. The condition is twofold: 1). Final landfill cover and the intermediate soil cover layers must first reach their field capacities so as to allow downward drainage of infiltration. 2). Also, the waste contained in each cell must be at field capacity. However, this has been an area of dispute for many researchers. If the above condition/s are satisfied, the percolation will seep downward and finally appear as leachate as it reaches the base of the landfill. These water balance components can be conglomerated, and expressed by the following equation:

$$PERC = P - RO - ET - AS + G \qquad \text{Eqn 2.1}$$

Wherein;
PERC= Percolation
P= Precipitation
RO= Runoff
ET= Evapotranspiration
AS= Storage capacity
G= Groundwater

In spite of this significant work, Farquhar (1989) noted that the above equation "is conceptual correct and comprehensive, accurate predictions of leachate flow are difficult to achieve because of the uncertainties associated with estimating the various terms. Most formulae and methods in use are empirical. Some of the data base required is stochastic in nature (temperature, heat index, precipitation, wind, vegetative). Other data are poorly defined (runoff coefficients, refuse cover density and compaction, moisture storage facilities)".

As a result, this has required more research to be conducted in order to significantly minimize the high degree of uncertainties/and or ambiguities associated with the water balance method. Apart from the study by Farquhar (1989), Blight et al. 1995; Blight and Blight, 1993; Blight, 2006, had carried out ongoing research to study the various components of a water balance in a sanitary landfill. The water balance for a landfill is defined in terms of the water input to the landfill, water output and water retained in the refuse body, and it applies the law conservation of mass.

Water input to landfill = Water output + Water retained in refuse body

And,

Water input includes: Precipitation (P)

The water content of the incoming waste (U_w)

NB. This variable makes a once-off account to the annual water balance to a given mass of landfill.

Water output includes: Evapotranspiration (ET)

Water lost in leachate escaping or removed from the landfill (L)

Runoff (R)

Lastly, the water absorbed and retained by the refuse (ΔU_w) and the soil cover (ΔU_s). The algebraic sum of an annual or seasonal water balance is as follows:

$$P + U_w = ET + L + R + \Delta U_w + \Delta U_s \qquad \text{Eqn 2.2}$$

Due to the lack of the information on the water balance components, Blight and Blight (1993) undertook a study to evaluate water balance components such as runoff and infiltration at Linbro Park landfill in Johannesburg, South Africa. The study was accomplished through simulation of rainfall events ranging through 5 mm, 10 mm and 20 mm depths, by means of using sprinkler infiltrometers. Their results revealed that most of the rainfall events typical of a summer rainfall area occur as low intensity events, and as result, the annual rainfall could almost entirely infiltrate the landfill surface. For this reason, they concluded that the low annual runoff could be prevalent in most climates and could be ignored with little error.

It has been further reported that for landfills which are no longer receiving waste, U_w in equation 2.2, as well as the runoff can be neglected. The water absorbed and retained by refuse (ΔU_w) and the soil cover (ΔU_s) were also found to have less significant effect on landfill water balance and as a result, they were both neglected. This has resulted into further simplification of equation 2.2, giving:

$$L = P - ET \ (\sim \Delta U_w) \qquad \text{Eqn 2.3}$$

Where;

L is the climatic water balance, and it is denoted as B in the Minimum Requirements

P is the annual rainfall (in mm)

ET is the evapotranspiration loss on the surface of a landfill (in mm)

It is noteworthy that the above equation (2.3) only comprises climatic variables and for this reason, it was for the first referred to as the climatic water balance in South African Minimum Requirements for waste disposal by landfill (Blight et al., 1995).

The present study is grounded on the climatic classification of the Minimum Requirements for disposal by landfill in South Africa. This classification has been widely used to evaluate the need for a collection and management system below a MSW landfill. It is based on the climatic water balance, and it uses published and easily available data obtained from the station nearest to the landfill. The climatic water balance is defined as: $B = R - E$, R is the annual rainfall (in mm) and E is the corresponding evaporation and evapotranspiration measured from the surface of landfill cover, both are measured in mm of water depth (Blight and Mbande, 1994; Blight, 2006). It is difficult to evaluate evapotranspiration (ET) from the surface of the landfill. A study by Hojem (1988) shows that: evapotranspiration (ET) = 0.7 × A-pan evaporation. This factor has been reported in two documents of the Minimum Requirement published in 1994 and 1998. However, a new climatic water balance method has been reported by Blight (2006), and it is now accepted that a 0.4 factor is more representative than the previously reported 0.70 factor, based on intensive field investigations of evapotranspiration.

The potential for a landfill to generate leachate is evaluated by the probability that the annual precipitation will exceed annual evapotranspiration. The evapotranspiration in a landfill is often assessed as a fraction of American standard A-pan evaporation. However, another type of pan called the Symons (S) pan is also used to measure evaporation and differs from the A-pan because of differing dimensions, conditions of exposure, as well as colour. Inevitably, these two types of pan provide slightly different results for evaporation losses (Blight, 1996). The A-pan evaporation apparatus is a circular tank, 1.18 m in diameter, 0.25 high deep and raised above the ground surface on a 0.15 high platform. The Symons pan is 1.83 m square, 0.61 deep and is sunk 0.54 m below the soil surface. Both pans are made of galvanized steel sheet (Blight et al., 1999).

The climatic water balance is calculated for the wet season of the wettest year on record and recalculated for successively drier years. Factors such as the moisture storage capacity of waste, surface runoff and capillary moisture are ignored as stated previously. If the value of B (climatic water balance) is calculated positive for less than one in five years, a site will classified as B^- (water deficient), no underliner or collection system is required due to no significant leachate generation. In the case where B is calculated as positive more than one in five years, a site is classified as B^+, and all these sites must be equipped with an underliner and leachate management system due to the significant potential for leachate generation (DWAF, 1998b). However, no study has been conducted for landfills situated on the borderline between B^+ and B^- climatic classifications. Figure 2.3 shows a dividing line for B^+ and B^- landfills in terms of the variables R and E in the climatic water balance equation $B = R - E$ where $E = 0.4 \times$ A-pan evaporation.

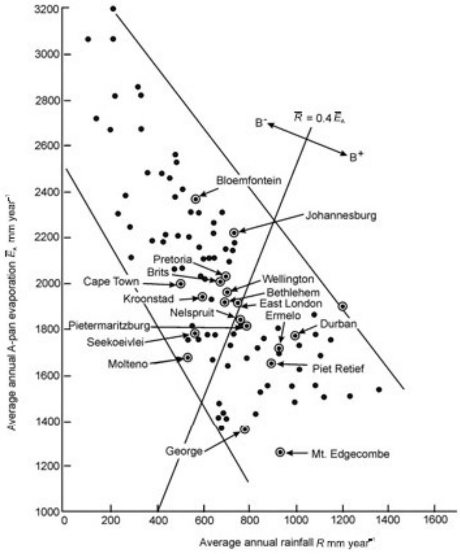


Figure 2.2: Relationship between average annual A-pan evaporation and average rainfall for several meteorological stations in South Africa (Blight, 2006).

2.4 SANITARY LANDFILL

The term landfilling originates from an operation of filling a hole in order to create a new land surface (Blight and Mbande, 1994). These authors pointed out that the science and engineering of municipal solid waste management is a relatively new concept. Fifty years back, a considerable amount of waste if not all, was disposed of in open dumps even by authorities serving large communities. The basic difference which separates a dump from a landfill is that in a dump there is no effort to separate the waste from the underlying soil or rock strata. With dumping, waste can be directly discarded into the groundwater if the hole extends beyond the groundwater level. To date, this practice is still common in most of the developing countries (Blight, 2010).

A turning point was reached in 1959 when the American Society of Civil Engineers (ASCE) recognized the definition of sanitary landfilling for the first time, more or less as it is known today (Blight and Mbande, 1994). Sanitary landfilling is an engineered method for disposing refuse or solid wastes by spreading the waste in thin layers to the smallest practical volume and applying and compacting soil to cover the waste at the end of each working day (ASCE, 1959, quoted by Shamrock 1998). It is an engineered method because different waste types are handled at a disposal facility that is designed, constructed and operated in a way of protecting public health and the environment.

The essential components of a sanitary landfill are those presented by Blight (1994). As depicted in figure 2.3, compacted wastes are separated by daily waste cover to form a cell. According to United States Environmental Agency (USEPA, 1984), the cell dimensions in a sanitary landfill are often determined by the volume of compacted waste. It is worthwhile to outline that the volume of compacted waste depends on the in-place refuse. The height of a cell is not restricted as long as the routine operations accommodate the placement of the cover material, as well as compaction on a daily basis. It is noteworthy that a sanitary landfill is constructed on a sealed impervious layer that is covered with a drainage system tailor-made to collect leachate. This layer significantly lessens the contamination of groundwater by leachate.

A network of leachate collection pipes make it possible for leachate generated at the base of a landfill to be easily transported to a designed point for treatment, in order to purify leachate prior discharging into a surface system.

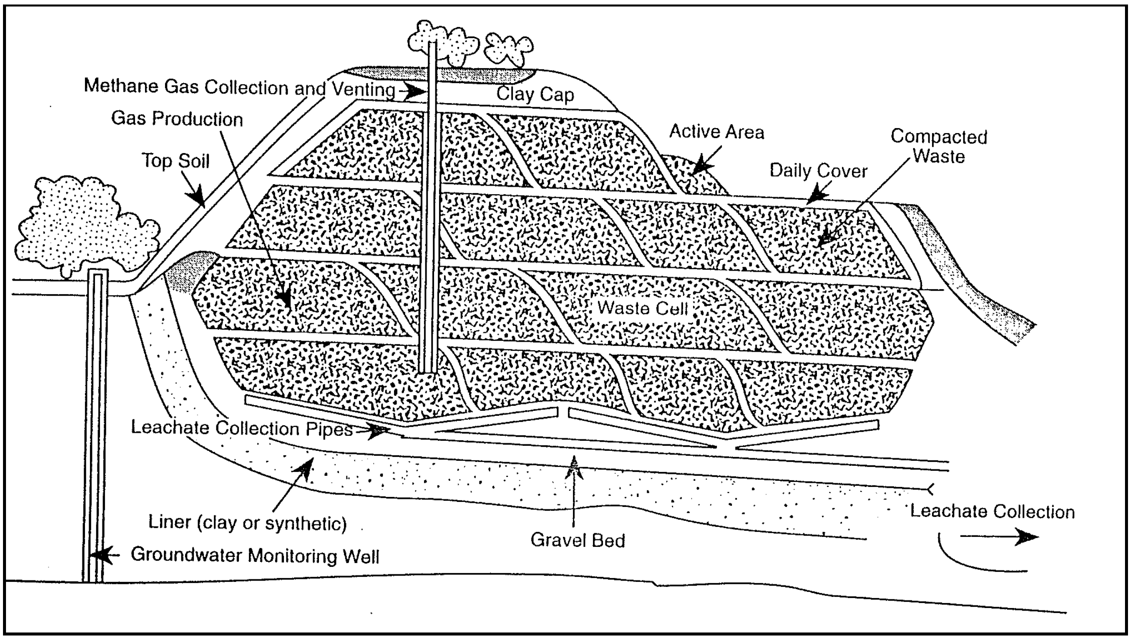


Figure 2.3: The essentials of a modern landfill (Blight, 1994).

Owing to the potential of methane gas production, a gas collection system is incorporated during the design of a sanitary landfill in order to collect and vent out methane gas. Leachate can deteriorate groundwater quality if it escapes at the base of a landfill; therefore, a borehole is often drilled in proximity to the landfill in order to allow continuous monitoring of groundwater quality.

In cases where a part of the landfill has reached its final stage of waste deposition, a specified cover material such as clay soil is recommended for final cover. The prime reason for using clay as a final cover is that the clay soils have low permeability; for this reason, it can reduce infiltration water entering the landfill, especially rainfall. Cover material in a landfill site can serve multiple purposes *viz.* minimization of fire hazards, odors, blowing litter, disease vectors such as rats and flies, control gas venting and water ingress from rainfall etc.

In a sanitary landfill, the size, type and number of pieces of equipment required collectively depends on the size and selected method of operation and to some extent on the experience and preference of the operators. There are two methods widely used for sanitary landfill practice, namely the area and trench fill method. However, sanitary landfills are operated as a hybrid of both the area and trench fill methods provided that extremely large amounts of waste must be disposed of (USEPA, 1984).

The area fill method. This method is also known as the progressive slope or ramp method. It involves construction of successive cells of waste that are compacted against a slope (figure 2.4). This method favors both even and moderately rugged terrains i.e. the area fill method can be used on flat or gently sloping land and also in quarries, strip mines, ravines, canyons, valleys etc. The mechanism of this method is that waste is spread and compacted on the surface of the underliner (if provided) and cover material is spread and compacted over the waste. The following sequence of operations is prevalent in a typical area fill method of sanitary landfilling:

- Construction of a starter berm or sloped back surface
- Construction of a liner and leachate collection system as required (With exception to South Africa, wherein landfills disposing of general waste under B^+ (water surplus) must be equipped with an underliner and leachate collection system)
- Deposit waste (MSW) at bottom of slope to achieve best compaction while controlling littering
- Spread and compact waste against slope of previous lift, progressing horizontally along slope
- Cover waste daily with earth materials from adjacent borrow area
- Cover final lift to a minimum depth of 500 mm
- Continue using progressive slope

It is, however, important to note that the aforementioned sequence of operations is not a standard; instead it is a cycle of operations and can vary from one place to another.

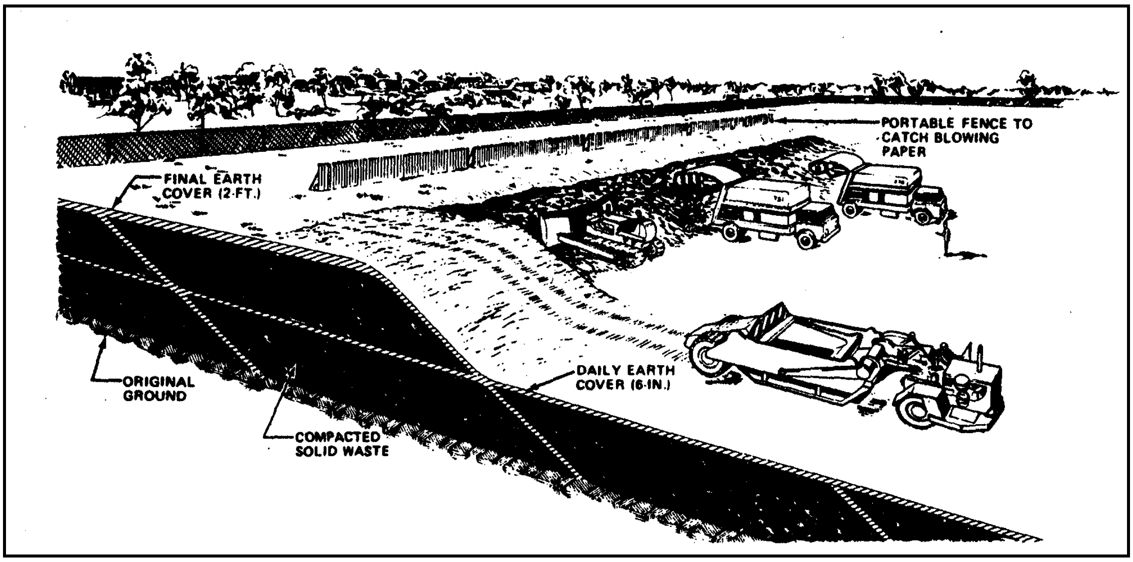


Figure 2.4: Area method.

The area fill method, like other methods has certain limitations. The strengths and limitations of the area fill method are tabulated in Table 2.3.

Strengths	Limitations
• Excavation of trenches is not required • Useful in those areas where the terrain may be unfavorable for trench operations • This method can accommodate high traffic volumes since the working face is not restricted by the size of operation	• Cover material may have to be imported • Greater litter problems is inevitable when applying this method • Higher operational costs due to greater amount of surface area utilized per volume of waste deposited • Topographic control not obvious as for trench method

Table 2.3: A summary of strengths and limitations of using the area fill method (Source: deq.state.wy.us/shwd/SW).

Trench fill method. From the word trench, this method is accomplished by means of spreading and compacting waste in an excavated trench. Unlike in area fill method; cover material for trench fill method is easily available as a result of the excavated trench (figure 2.5). The undesired spoil material for daily cover may be stockpiled and used at a later stage as a cover for area fill method designed for the top of the completed trench fill operation. The inherent nature of cohesive soils such as till or clayey silt is more desirable for use in trench fill operation because the walls separating the trenches can be thin and nearly vertical. Weather conditions and lengthier duration at which the trench has to remain open can drastically affect the soil stability and must be considered during the design of the slope of trench walls. The amount of blowing litter can be significantly minimized if the trenches are oriented perpendicularly to the prevailing blowing wind (USEPA, 1984).

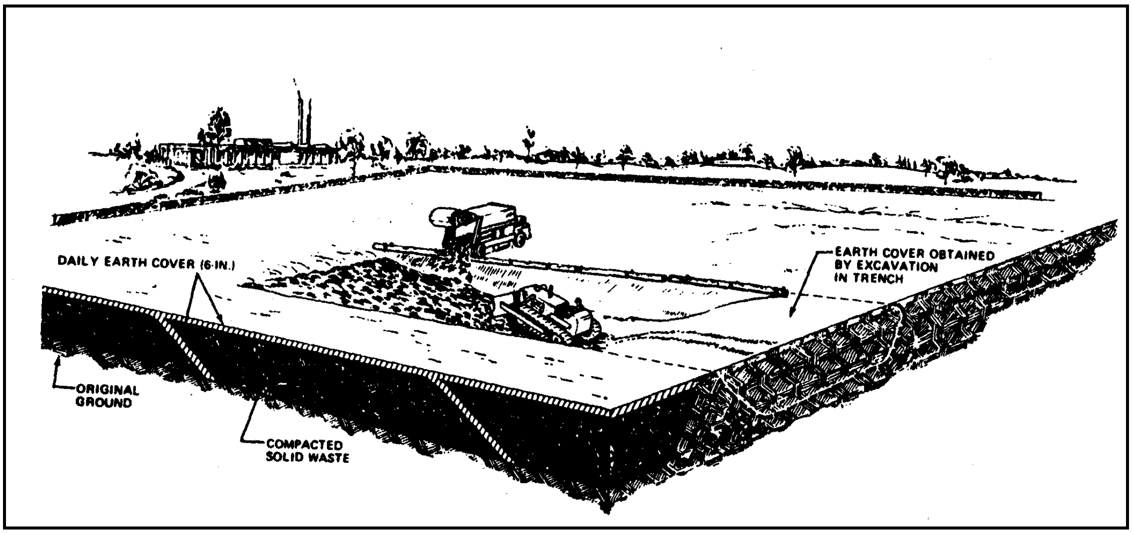


Figure 2.5: Trench fill method.

Table 2.4 summarizes the strength and limitations of the area fill method of deposition.

Strengths	Limitations
• Sufficient cover materials and therefore, it's not necessary to import cover material • Operation can be achieved with a minimum working face exposed • Can be designed for optimum drainage during filling operations • Blowing litter can be easily controlled with this method • Lower area/volume ratio may reduce leachate production potential	• Leachate collection system may be more difficult or expensive to install • Loss of landfill volume due to walls separating the trenches • Trench depths together with side slopes depend on soil type and stability • Trench size may not be sufficient to handle high traffic volumes of waste

Table 2.4: A summary of strengths and limitations of using the area fill method (Source: deq.state.wy.us/shwd/SW).

2.5 BIOREACTOR LANDFILL

According to the Solid Waste Association of North America (SWANA), a bioreactor landfill is:
"a controlled landfill or landfill cell where liquid and gas collection are actively managed in order accelerate or enhance biostabilization of the waste. The bioreactor landfill significantly increases the extent of organic waste decomposition, conversion rates, and process effectiveness over what would otherwise occur with the landfill".

A study by Reinhart et al., 2002 indicates that literature references have increased dramatically since the inception of this technology. The prime reason for application of bioreactor technology is to attain refuse stabilization in a sanitary landfill rapidly. Stabilization in a landfill may be defined as "a process or series of processes producing an end product of such characteristics that its ultimate use will be acceptable in terms of both environmental and public health'' (Ross, 1990). According to Hughes and Christy (2003), stabilization may never occur in a conventional landfill, or it may take approximately 100 years to take place. Hence, in a bioreactor landfill stabilization should occur within ten years or even less. However, there is reluctance to apply bioreactor technology due to the fact that the technology is not well demonstrated and there are technical impediments, unclear cost implications, and regulatory constraints (Reinhart et al., 2002).

The difference which marks bioreactor landfill from sanitary landfill is that in a bioreactor landfill, the amount of leachate generated at the bottom of a landfill is recirculated back into the landfill in a controlled manner, while there is no leachate recirculation taking place in a sanitary landfill (see figure 2.6). Research shows that leachate recirculation accelerates refuse degradation due to the continual flow of leachate through the waste. However, leachate recirculation may lead to instability of the landfill, for example, the failure in Dona Juana landfill in Bogota, Colombia in 1997 (Blight, 2008). In this method, the degradation of refuse is promoted by adding certain elements (nutrients, oxygen or moisture) and controlling other parameters such as pH and temperature. A summary of advantage and disadvantages of bioreactor landfill is provided below.

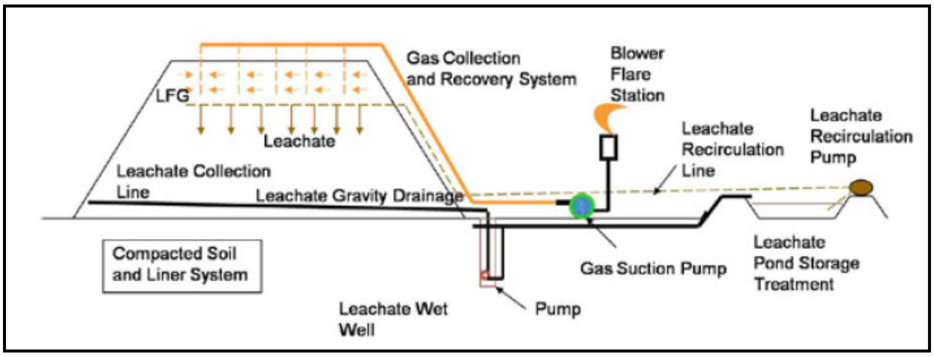


Figure 2.6: The major components of a bioreactor landfill (Townsend et al., 2008).

Advantages	Disadvantages
• Rapid waste stabilization • Improved gas emissions • Degradation of recalcitrant chemicals • Removal of moisture	• Risk of fire and explosion of gas mixtures • Cost • Unknown gas emissions

Table 2.6: A summary of advantages and disadvantages of a bioreactor landfill (Reinhart et al., 2002).

2.6 LANDFILL PROCESSES

2.6.1 Leachate generation

The common notion is that leachate will be generated once the refuse has reached its field capacity. By definition, field capacity is the maximum amount of moisture that a porous medium can retain without downward percolating seepage (El-fadel, 1997). The effect of field storage capacity on leachate production has been an area of interest for many researchers (e.g. Harris, 1979; Holmes 1983; Blakely, 1992 and Blight et al., 1992). The physical characteristics of waste are important in order to understand some of the landfill processes such as leachate generation. Bengtsson (1994) noted that the inherent nature of waste contained within a landfill is a constituent of small-scale spatial heterogeneities resulting from the presence of large and more or less continuous voids. These small-scale spatial variations found in waste within a landfill are referred to as macropores. Furthermore, macropores are typically characterized by higher hydraulic conductivity as compared to the surrounding waste matrix. Blight et al. (1992) attributed the generation of leachate from waste below its overall field capacity to the effects of channeling flow through macropores.

A study by El-fadel et al. 1997 outlined major factors influencing leachate formation. These factors include: climatic and hydrogeologic, site operations and management, refuse characteristics and the internal landfill processes. Moreover, these factors are further subdivided into those directly related to landfill moisture content and those which affect the distribution of moisture content within the landfill (see figure 2.7). Rees (1980) also pointed out four key factors affecting leachate production in typical landfill conditions:

1. The water content of waste when placed
2. The volume of rainfall or other water allowed to enter the waste
3. The volume of liquids or sludges co-disposed with the refuse
4. Waste compaction and density

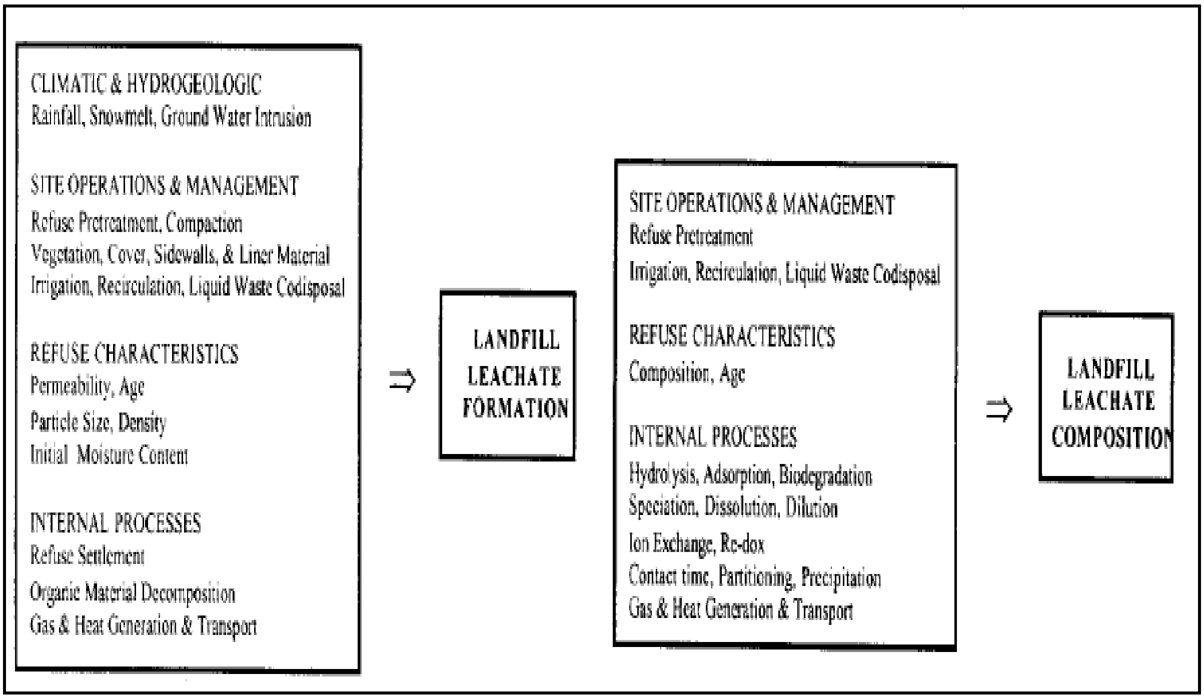


Figure 2.7: Factors influencing leachate formation and composition (El-Fadel et al., 1997).

1. The water content of waste when placed

The water content of the incoming waste is a factor independent to the climate in regard to the leachate generation (Blight and Fourie, 2005). These authors emphasized that incoming waste carries relatively high moisture content even in areas situated in arid or semi-arid environments. The high moisture content of the incoming waste is attributed to the presence of a high fraction of organic or putrescible waste. Table 2.6 shows 32 % of putrescible waste for Cape Town and 45 % for Johannesburg. In a separate study by Blight et al. 1992, the moisture content results for Cape Town, when expressed as a percentage of air-dry mass, varied from 18 to 37 %, whereas for Johannesburg it varied from 30 to 50 %. If the moisture content results for both cities (Cape Town and Johannesburg) are to be correlated to the corresponding percentage of putrescible waste for Cape Town and Johannesburg as depicted in Table 2.6 below, it becomes apparent that the moisture content increases with the putrescible fraction. Landfills situated in desert areas such as Gaborone, Lima and Abu Dhabi are also characterized by high moisture content because of the higher percentage of putrescible waste they receive. The increase in moisture content is the major contributor to leachate generation (El-fadel, 1997).

Component	USA	UK	Cape Town	Johannesburg	Gaborone	Lima	Delhi	Abu Dhabi
	% Composition by undried mass							
Ash, soil, building rubble, other	8–14	18	39*	1**	3	16	41	16
Paper and cardboard	34–44	29	19	23	12	14	9	6
Plastic	8–10	7	7+	13	5	7	1+	12
Metals	11–13	8	2+	10	6	4	1+	8
Glass	4–9	10	1+	8	6	3	1+	9
Organics, putrescibles	12–22	25	32	45	68	56	47	49

* Building rubble landfilled, includes sand cover. ** Building rubble not landfilled, cover soil not included. + Scavenged before landfilling.

Table 2.6: Composition of waste from USA and UK compared with waste from Cape Town, Johannesburg, Gaborone (Botswana), Lima (Peru), Delhi (India) and Abu Dhabi (United Arab Emirates) (After, Blight and Fourie, 2005).

2. The volume of rainfall or other water allowed entering the site

The generation of leachate in most parts of the world is mainly influenced by the climate. South Africa is typically characterized by semi-arid and arid climate where the potential evaporation or evapotranspiration exceeds the annual rainfall (Blight and Fourie, 2005). Owing to the perennial water deficit is such areas, leachate is only sporadically generated after unusually wet weather. Thus, the South Africa Minimum Requirements for waste disposal by landfill make use of the climatic water balance to evaluate the potential for a landfill to generate leachate or not.

Precipitation is the climatic variable which accounts for highest percentage of incoming moisture in a landfill (Rohrs, 2002). Precipitation on a landfill site can occur in various forms such as rain, hail, sleet and snow, but rainfall predominates in temperate to hot climates. The percolation of precipitation into a landfill surface, especially rainfall can be affected by several factors such as rainfall intensity and duration, antecedent soil moisture conditions, site slope, permeability and vegetation cover. These factors collectively affect the proportion of the rain which runs off the landfill; a steep slope angle coupled with low hydraulic conductivity soil covers such as clay retards the downward percolation seepage (Rohrs, 2002). A high rate of infiltration is favored by light rainfall over a prolonged period of time, while short bursts of heavy rainfall may result in early saturation of soil cover with the remaining amount being lost as runoff (Scott et al., 2005).

According to Scott et al. (2005) most of the moisture entering the landfill is primarily derived from the infiltration of rainfall, however, the surface run-on from around the landfill and groundwater may account for some percentage of water entering the landfill. The generation of leachate in a sanitary landfill can be directly influenced by groundwater (Farquhar, 1989). Landfills situated in areas of shallow water table can produce substantial amounts of leachate if the water table can rise up into the landfill. For landfills situated in steep sided topography in water deficient sites, where leachate is unlikely to be generated, Blight (personal communication) noted that run-on water may contribute to a high leachate generation rate if the site is not equipped with a drainage interception system upslope.

3. The volumes of liquids or sludges co-disposed with the waste

Al-Yaqout and Hamoda (2003) undertook a study to evaluate landfill leachate in arid climates. The common sense is that landfills situated in arid regions cannot generate leachate. However, considerable amounts of leachate were detected in landfills situated in arid climates. They attributed the primary sources of leachate generation to the improper disposal of liquid and sludge coupled with rising of the groundwater table. Thus, the Minimum Requirements for waste disposal by landfill in South Africa enforce that the co-disposal ratios need to be calculated with high level of confidence prior waste-liquid and sludge co-disposal (Blight, 2006). Liquid squeezed out of other waste, e.g. food and garden waste (known as "squeezate") may also add to overall leachate generation.

4. Waste composition and density

Blight and Fourie (2005) noted that the waste rich in putrescible matter can be easily compressed by increasing the overburden. As a result, this can lead to early generation of leachate as squeezate, and squeezate alone can pose more serious pollution problems than rain infiltration-induced leachate. Therefore, compaction density is one of the important aspects to be dealt with when disposing of waste of varying moisture content. In addition, less compacted or loose waste may promote generation of high volumes of leachate, since compacted waste reduces the infiltration rate.

Other factors influencing leachate generation

Hydrogeology

Landfills around the world rest on various geological terrains. Owing to the difference in the properties of different subsurface geological formations, some rocks and soils exhibit high permeability. As a result, rocks that act as groundwater aquifers may determine the fate of any leachate released together with its contaminants (Scott et al., 2005). The water bearing formation is typically characterized by high porosity and permeability and for this reason, water can readily move towards the landfill site to augment the leachate generation. The soil type beneath a landfill site can affect the generation of leachate, notably, sandy soil because of its high porosity, while clay layers characterized by small pore spaces can prevent the easy movement of groundwater. Inevitably, the low hydraulic conductivity of soils containing high proportions of clays is the prime reason they are used to line and cap landfill sites.

Final landfill cover

Most of the regulations governing the disposal of municipal waste require all landfills to be covered with specified soil materials. The principal reason is to eliminate water ingress into a landfill, and if it is successful, leachate generation can presumably be minimized. Research has shown that soils with low hydraulic conductivity such as clays are suitable for landfill cover. However, several studies including a study by Ham and Bookter (1982) noted that cracks and fissures can develop in a landfill soil cover, and these facture zones can act as corridors for fluids to reach the waste body. Furthermore, lateral flow within the soil cover towards a more permeable downhill zone is also possible.

Karnchawong and Yongpisalpop (2009) carried out a study to investigate the effects of using different types of soil cover layers on leachate generation and composition from four landfill lysimeters in Thailand. In their study, three of four lysimeters were filled with municipal solid waste and capped with different cover soil types such as sandy loam, silt loam and clay soil while the remaining lysimeter was only filled with municipal solid waste. The cumulative leachate quantity measurements from four lysimeters revealed that the lysimeter filled only with MSW produced approximately

27 % more of leachate than the three lysimeters capped with different soil covers. There were no clear trends of the cumulative leachate amounts generated from the three lysimeters despite their difference in soil covers. Alternatively, vegetation can be used as part of a final landfill cover. Vegetation has the capability to intercept precipitation prior to reaching ground surface and re-direct some of the infiltration into the atmosphere through the process of evapotranspiration (Blight et al., 1995). So, bare soil covers on a landfill may result in more leachate production than vegetated covers.

Age of a landfill

Leachate generation can also be influenced by the age of landfilled waste. A water balance study for landfills of different ages in Sweden showed that refuse age can have an effect on landfill leachate generation (Bengtsson et al., 1994). Smaller quantities of leachate were noted in young landfills than in old landfills. This can be explained in terms of greater retention of water in the fresh refuse; the fresh landfilled wastes were characterized by macropores and high field capacity. In case of old landfills, the field capacity was found to be low; hence, a slow, more continuous seepage of leachate was noted.

2.6.2 Leachate composition

An understanding of leachate composition is critical for making projections of long term adverse environmental impacts of landfilling. It is difficult to characterize leachate because its composition and concentrations depend on a variety of factors such as waste composition, landfilling technology, waste age, state of closure, geology, temperature, moisture content and other seasonal and hydrological factors (Kjeldsen et al., 2002). However, the quality of leachate generated is primarily dependant on the balance between the acetogenic and methanogenic phases of refuse generation (Shamrock 1998). Other factors influencing leachate composition include particle size, degree of compaction, climate, etc. (Lema et al., 1988).

Refuse composition depends on the standard of living of the surrounding population. A study by Shamrock (1998) revealed that rich wastes are characterized by higher pollutants loads than poor waste. This was evident in higher concentrations of chloride, ammonia, nickel and potassium in lysimeters filled with rich waste only. However, there was less difference in COD levels in lysimeters containing poor and rich wastes than expected. The slight difference was attributed to the type of test method used for analysis during the course of the study. In addition, leachate sampling methods and sample handling can further affect the chemical properties of leachate (Kjeldsen et al. 2002; Lema et al., 1988).

The age of the tip or landfill is one of the prime factors which influence leachate composition and it mirrors the degree of stabilization of the refuse. In the early stages after the burial of refuse, acetogenic leachates exhibit high values of BOD and COD and increasing concentrations of ammonia, while these organic compounds are converted into landfill gas in later methanogenic phases of refuse decomposition (Kjeldsen et al. 2002). Robinson and Gronow (1993) define acetogenic leachate as having concentrations of volatile fatty acids greater than 1000 mg/l, with COD values ranging from 20 000 to 40 000 mg/l. Owing to production of simple organic acids and dissolved CO_2 in the leachate, this stage is marked by low pH values (Ehrig, 1991). As a result, acetogenic leachate is characterized by high concentrations of heavy metals. In case of methanogenic leachate, the concentration of volatile fatty acids is less than 200 mg/l and chloride values may be over 500 mg/l (Robinson and Gronow, 1993).

This leachate has been widely referred to as stabilized (e.g. Gettinby et al., 1996). Ehrig and Scheelhaase (1993) indicated that methanogenic leachate is characterized by high pH values and a BOD_5: COD ratio of less than 0.1, while acetogenic leachate is characterized by a BOD_5: COD ratio of 0.4. In conclusion, young landfills are characterized by leachate with high organic strength and may pose adverse environmental impacts, whereas old landfills may pose little or no threats because of the weakened chemical aggressiveness of leachate as the landfill ages.

Leachate composition is also affected by climate, which varies from one locality to another. Several studies on the effect of climate on leachate composition show the seasonal variations in leachate composition. For instance, the concentration of pollutants is lower during rainy season (Kjeldsen et al., 2002), when leachate is more diluted.

Several parameters are used to describe the contaminant concentrations in leachate. These parameters include chemical oxygen demand (COD), Ammonia Nitrogen (NH^+_4-N), Dissolved Solids (DS), Suspended Solids (SS), Xenobiotic Organic Compounds (XOCs), heavy metals, salts etc. (Ziyang 2009). A study by Kjeldsen et al. 2002 grouped pollutants found in municipal solid waste (MSW) landfill leachate into three main groups.

- Dissolved organic matter, quantified as Chemical Oxygen Demand (COD), Biological Oxygen Demand (BOD) or Total Organic Carbon (TOC) and volatile fatty acids.
- Inorganic macrocomponents: calcium (Ca^{2+}), magnesium (Mg^{2+}), Sodium (Na^+), potassium (K^+), ammonium (NH_4^+), iron (Fe^{2+}), manganese (Mn^{2+}), chloride (Cl^-), sulfate (SO_4^{2-}) and hydrogen carbonate (HCO^{3-}).
- Heavy metals: cadmium (Cd^{2+}), chromium (Cr^{3+}), copper (Cu^{2+}), lead (Pb^{2+}), nickel (Ni^{2+}) and zinc (Zn^{2+}).

2.6.3 Landfill gas production

Several studies have been conducted to gain insight into landfill gas-forming processes, specifically methane production (Farquhar and Rovers, 1973; Pohland, 1975; Rees, 1980; Ramaswamy, 1970; Klink and Ham, 1982; Hartz and Ham, 1983; Bogner and Spikos, 1993; Bogner, 1992). Farquhar and Rovers (1973) undertook a study to investigate gas production during refuse decomposition. In their study, the primary aim was to investigate the pattern of gas production in conjunction with the factors affecting that pattern. A group of factors were identified and investigated i.e. group A, B and C factors as shown in figure 2.8. The group 'A' factors, viz. temperature, aeration, moisture content, Eh, pH, alkalinity, nutrition and toxic compounds, are features of the immediate microbial environment under which gas production takes place. These factors are discussed in detail in section 2.6.4.

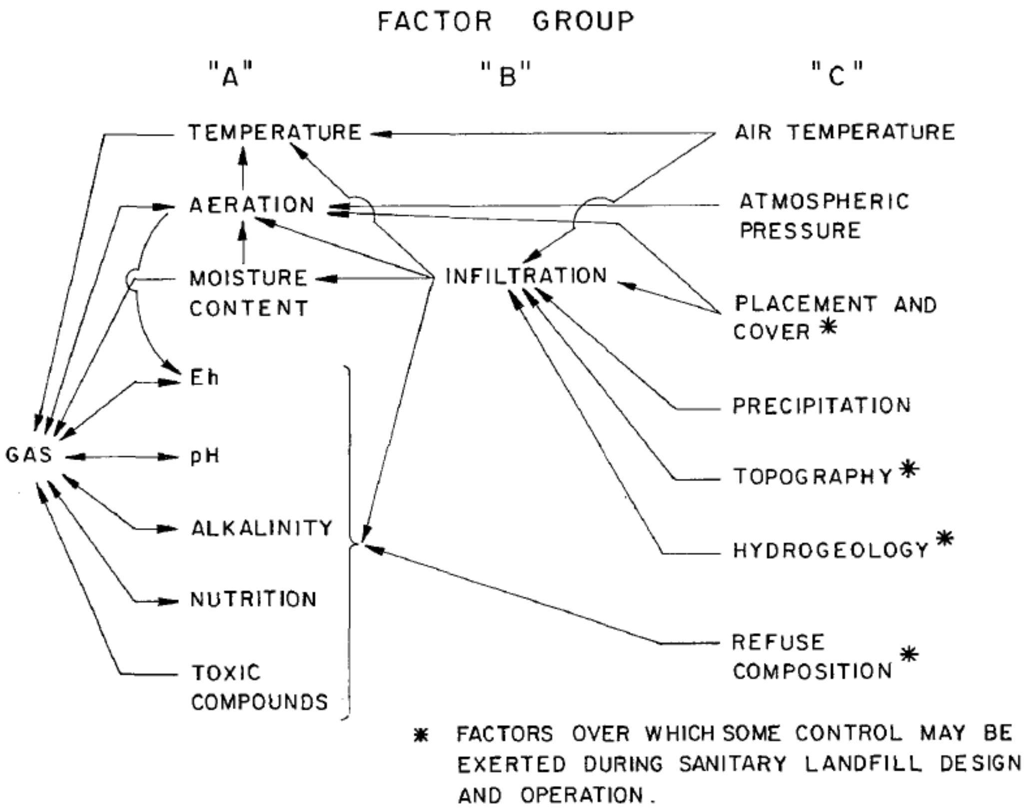


Figure 2.8: Interactive factors affecting gas production in sanitary landfill (After, Farquhar and Rovers, 1973).

The Group 'B' factor, namely infiltration is most important, and this factor appears to have a severe impact on group 'A' factors (see figure 2.8). However, its severity of impact is governed by the amount of water infiltrating the landfill and its chemical physical characteristics. The group 'C' factors include air temperature, atmospheric pressure, placement and cover, precipitation, topography, hydrogeology and refuse composition, many of which affect infiltration.

The effect of moisture content on landfill gas production is well documented by several researchers (e.g. Farquhar and Rovers, 1973; Pohland, 1975; Rees, 1980; Ramaswamy, 1970; Klink and Ham, 1982; Hartz and Ham, 1983; Bogner and Spikos, 1993; Bogner, 1992; Wreford, 2000). Most of the studies conducted have attributed an increase in gas production to increased moisture content. A study by Bogner (1992) has reinforced such evidence by studying the effects of moisture condition on biogas (methane) production (see figure 2.9 below). Ramaswamy (1970) pointed out that maximum methane (CH_4) production can be attained when the refuse moisture content is in the range of 60 to 80 % by wet weight. Farquhar and Rovers (1973) noted that at moisture content levels more than 80 % (on a total weight basis) methane production decreases. According to Hartz and Ham (1983) at least 10 % of moisture content by wet weight is needed in order to produce methane.

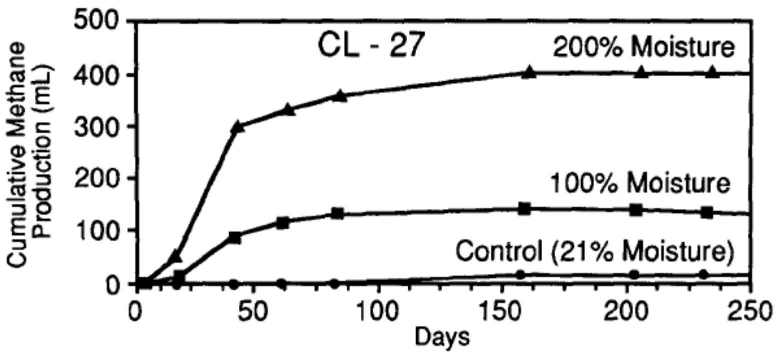


Figure 2.9 Effect of moisture condition on biogas production. Typical plot of cumulative methane production (mL) from in-vitro anaerobic incubation of 25-g landfill samples with gravimetric water content adjusted to 100 and 200 % (After, Bogner, 1992).

To date, the fact that a higher moisture content influences the production of methane is universally accepted. At optimum moisture contents, the contact among microorganisms, nutrient, and degradable substrates is enhanced, and microbial metabolism is also accelerated (Bogner and Spikos, 1993). However, air ingress during excessive infiltration can inhibit methane generation.

Higher infiltration, however, can also inhibit methane production. Farquhar and Rovers (1973) undertook a study to evaluate gas and methane production in refuse placed in a 400 cubic foot (11.32 m^3) cylindrical test cell. At day 115 after the placement, a concentration of 19 % by volume of methane was attained. The continuous melting of snow and ice led to infiltration of large amounts of water into the refuse, and inevitably large volumes of leachate were generated. As a result, a decline in methane gas production from 19 % to 4 % by volume was detected just after the high infiltration had taken place. Furthermore, increased concentration in leachate quality parameters such as chemical oxygen demand (COD), biochemical oxygen demand (BOD) and volatile dissolved concentrations (VDS) was noted, while alkalinity and pH decreased. The maximum pH for landfill gas production is 7.0, while a minimum is 5.5 (Farquhar and Rovers, 1973).

To investigate the advantages of a high moisture content for methane production, Buivid *et al.,* 1981 studied the relationship which exists on landfill enhancement parameters, and their effects. Their results revealed that the moisture content alone does not enhance methane production, but when added to nutrient, sludge and buffers enhances the degradation process. Nutrient in the form of sewage sludge are sometimes used to accelerate methane production in a sanitary landfill. It is envisaged that once the distribution of moisture and nutrients has been achieved, additional moisture is less significant (Buivid et al. 1981, quoted by Komilis 1999). Cossu et al. (1987) noted that MSW co-disposal with digested sewage sludge and fertilizer (Nitrogen and Phosphorus; nutrient) with under saturated moisture conditions results in increased gas production. On the other hand, refuse (MSW) only, saturated with water produced less gas. In addition, the anaerobically digested sewage sludge can triple methane production above that of a mixture of MSW with primary sludge. The possible reason is the presence of higher populations of methanogenic bacteria and/or a nutrient deficit in primary sludge.

Leachate recirculation can also enhance the production of methane from refuse (Pohland, 1975). Barlaz et al. (1987) however noted that leachate recycling alone does not enhance methane production, unless neutralized and added with buffers.

Temperature is one of the significant factors likely to play a crucial role in regulating gas production rates in sanitary landfills (Farquhar and Rovers, 1973). Different temperature ranges in which the methanogens operate include pychrophilic (< 20 ^{0}C), mesophilic (20 - 45 ^{0}C) and thermorphilic (> 45^0) (Boyd, 1988).

The mesophilic bacteria population which is responsible for the gas production in a landfill will decline if it is perpetually exposed to temperatures above 40 ^{0}C (Britz and Tracey, 1990). According to Farquhar and Rovers 1973, the optimum temperatures for gas production are in the range of 30 to 35 ^{0}C, and it is imperative to maintain the lysimeter temperature in this range because methanogenic bacteria thrive within the specific temperature ranges.

Attempts to operate landfills as bioreactors have received attention for the past decade. Inevitably, this has permitted the landfill processes such as waste decomposition and methane production to be enhanced (Komilis et al., 1999). The principal reason of operating a landfill as a bioreactor is to shorten the acidogenic phase, therefore, reducing leachable organic emissions and enhancing methane production.

The coal ash (pulverized fuel ash) from electricity power plants and MSW incinerator bottom ash co-disposed with MSW were found to have beneficial effects on waste decomposition via an increase in gas production (Cossu et al., 1991). Their results indicated that 10 % by weight of coal ash co-disposed with MSW resulted in no inhibition of the methanogenic phase and accelerated degradation. The coal ash co-disposal with MSW can result in the rapid establishment of methanogenic conditions by neutralizing acids formed during the acidogenic stage of anaerobiosis, and this justifies its use in the present study.

2.6.4 Waste decomposition

In chapter one (section 1.4) , it was mentioned that the present study sets out to investigate generation of leachate from landfills with climatic classifications between B^+ and B^- and decomposition of municipal solid waste under these conditions. The former was discussed in detail in this chapter (section 2.6.1) and the latter will be comprehensively discussed under this section. It is important to note that the literature survey presented in this study for both leachate generation and municipal solid waste degradation mostly involves studies conducted in wet (B^+) landfills in different parts of the world. Until recently, there has been no research conducted to understand refuse degradation processes for landfills in South Africa situated on the boundary between B^+ and B^- . It is the aim of the present study to add valuable information on the current existing knowledge pertaining to waste decomposition.

A lot of research has been conducted in order to gain knowledge on the degradation processes of MSW which occur in typical landfill environments (e.g. Farquhar and Rovers, 1973; Rees 1980; Pohland and Engelbrecht, 1976, Tchobanoglous et al., 1993). This was mainly achieved through the in-depth examination of the major end-products such as methane (CH_4), carbon dioxide (CO_4) and volatile fatty acids that a landfill emits during refuse degradation. A study by Tchobanoglous et al., 1993 indicated that MSW degradation is a function of its physical, chemical and biological properties. Notably, biological transformation processes are of more importance than physical and chemical properties.

Physical transformation processes

Physical properties of MSW include specific weight, moisture, particle size distribution, field capacity, porosity and compatibility (Morris, 2001). According to Tchobanoglous et al. 1993, the physical transformation processes available for management of solid waste are component separation, mechanical volume reduction, compaction and particle reduction (shredding) prior to disposal of MSW to landfill.

Chemical transformation processes

The chemical transformation processes in a landfill occur in the various form of chemical reactions such as precipitation, adsorption, desorption and dissolution (Blight, 1992). Most importantly, chemical properties of MSW such as energy potential and nutrient availability are highly affected by refuse composition.

Biological transformation processes

Biological transformation processes in typical landfill conditions predominate over both physical and chemical transformation processes and play a significant role in refuse degradation. Biological properties of MSW include various proportions of volatile acids, sugars, fats, hemicellulose, cellulose, proteins and lignin and hence affect its biodegradability (Tchobanoglous et al., 1993). Biological transformation of the organic fraction of refuse (MSW) may be used for a variety of purposes such as reduction of volume and mass of the material in order to produce compost or methane. Biological processes are either aerobic or anaerobic depending on their demand for oxygen.

Aerobic degradation. This process is primarily used for the conversion of organic matter by means of aerobic bacteria in order to produce compost. In this process, there are numerous end products which follow aerobic degradation and are shown by the following reaction (Tchobanoglous et al., 1993; Morris, 2001; Shamrock, 1998):

$$\text{Organic matter} + O_2 \rightarrow$$
$$\text{New biomass} + \text{Resistant organic matter (compost)} + CO_2 + H_2O + NH_3 + SO_2^{2-} + \text{Heat}$$

Aerobic degradation of refuse is time dependent and is also influenced by the nature of the waste, moisture content, nutrients availability and other environmental-related factors.

Anaerobic degradation. Anaerobic degradation of organic matter is often complex to understand and has been described by many researchers (e.g. Farquhar and Rovers, 1973; Rees 1980; Ham and Bookter, 1982; Evans, 2001; Kjeldsen et al., 2002). The anaerobic degradation of the organic fraction of MSW is followed by production of gas such as CO_2 and methane which together constitute over 99 % of total gas produced (Tchobanoglous et al., 1993). The following reaction shows the end products during anaerobic degradation:

$$\text{Organic matter} + H_2O + \text{Nutrients} \rightarrow$$
$$\text{New biomass} + \text{Resistant organic matter} + CO_2 + CH_4 + NH_3 + H_2S + \text{Heat}$$

There are four main distinct steps of refuse decomposition and each of which is characterized by specific groups of microorganisms (Morris, 2001). Figure 2.10 depicts basic steps of the anaerobic degradation process. The steps of refuse decomposition together with three main groups of microorganisms are summarized below.

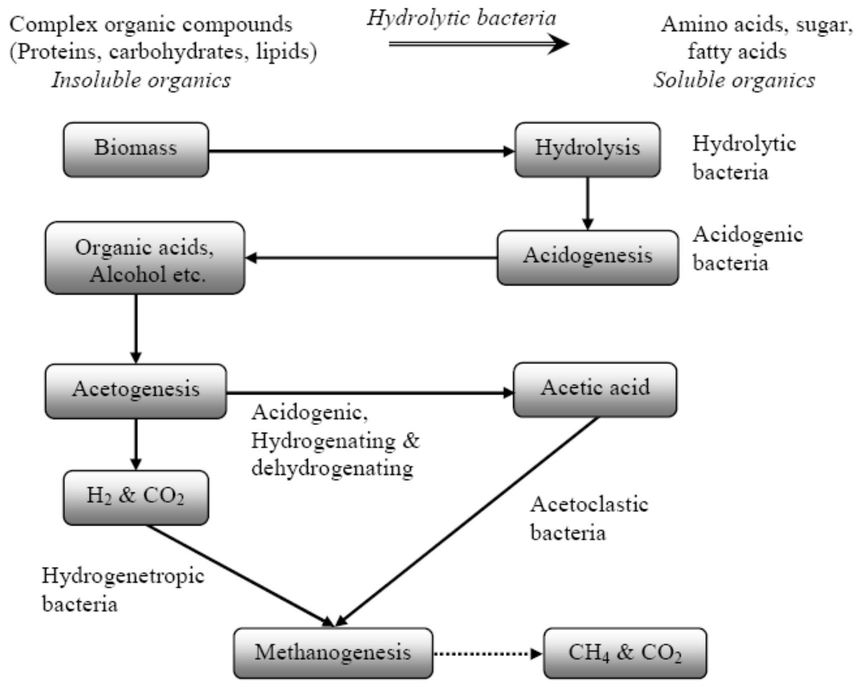


Figure 2.10 Anaerobic degradation process (Evans, 2001)

Four main steps of refuse degradation

1. Hydrolysis

During hydrolysis the complex solid and dissolved organic materials are broken down into smaller, soluble components required for subsequent microbial conversions (Morris, 2001). The hydrolytic organisms responsible for hydrolysis process (hydrolyze) - organic matter such as proteins, poly carbonates, lipids, etc. to form simple organic compounds such as formate, acetate, propionate, butyrate and other fatty acids (Chaudhary, 2008). There are several factors influencing the rate of the hydrolysis process and these include pH, temperature, concentration of hydrolytic bacteria, and type of particulate organic matter (Neves et al., 2006). According to Themelis (2004), the following reaction takes place when the organic fraction of refuse (MSW) is broken down into a simple sugar (glucose):

$$C_6H_{10}O_4 + 2H_2O \rightarrow C_6H_{12}O_6 + 2H_2$$

2. Acidogenesis/Acid fermentation

During Acidogenesis/Acid fermentation the hydrolyzed compounds are fermented to volatile fatty acids (acetic, propionic, butyric, valeric acids, etc.), alcohols (ethanol, methanol), hydrogen, ammonia and CO_2 (Chaudhary, 2008; Morris, 2001). This stage is marked by low pH levels especially when these compounds increase. Two groups of microorganisms responsible for this biological transformation are anaerobes and facultative bacteria.

3. Acetogenesis

Acetogenesis is the process whereby long chains of volatile fatty acids (VFAs) produced during acidogenesis are further converted into acetate (the shortest chain VFA), hydrogen and CO_2 (Morris, 2001).

4. Methanogenesis

Methanogenesis is the last stage in the list of anaerobic degradation of refuse steps. In this process, the methanogens which are bacterial populations responsible for methane production are divided into two groups, namely acetoclastic and hydrogenotrophic methanogens. In acetoclastic methanogenesis, acetic acid is converted to form methane and CO_2, while hydrogenotrophic group consume hydrogen and CO_2 to form methane (Evans 2001; Morris, 2001).

Phases of waste decomposition

It is accepted that a landfill must undergo at least four phases in the process of stabilization (Kjeldsen et al., 2002; Yuen, 2001; Barlaz and Ham, 1993; Morris, 2001). The time for a particular phase to occur depends on a number of factors such as nutrient availability, waste moisture content and infiltration rates, routes and compaction etc (Morris 2001). Figure 2.11 (Saint-Fort, 1992) shows five distinct phases that a landfill must go through in the process of stabilization. These include initial adjustment phase, transition phase, acid formation phase, methane fermentation and final maturation phase.

Phase I. This is an initial adjustment phase which is marked by the presence of oxygen contained in the voids of freshly placed refuse. The rapid consumption of oxygen contained within refuse voids is followed by the production of CO_2 and an increase in internal temperature. This phase lasts for a couple of days since oxygen is not replenished once the waste is landfilled. Most of the leachate produced during this phase results from the release of moisture during compaction. Generally, during the initial adjustment phase, the waste is not typically at field capacity.

Phase II. Owing to the depletion of oxygen which occurs during the initial adjustment phase, the waste becomes anaerobic. In this phase, namely transition phase, fermentation reactions are supported. Initially, nitrate and Sulphate are reduced to nitrogen gas and hydrogen sulphide. As the redox potential drops, the microbial community responsible for the production of hydrogen, CO_2 and methane begin the three step process of hydrolysis, acidogenesis and methanogenesis. The elevated CO_2 concentrations and production of VFAs cause a drop in pH.

Phase III. During this, the acid formation phase, microbial activity initiated in Phase II accelerates with the production of significant quantities of VFAs and lesser amounts of hydrogen from the first two steps of the three step process (hydrolysis and acidogenesis). CO_2 is initially the main gaseous product. The pH of leachate, if formed, often drops to a value below 5 and the initial concentrations of sulphates are slowly reduced as the redox potential drops. The biochemical and chemical oxygen demands (BOD and COD) increase significantly due to dissolution of organic acids in leachate. Ammonia concentrations in leachate are typically high. The low pH of acid causes a number of inorganic constituents of the MSW, principally heavy metals, to be solubilised. Many essential nutrients are also removed in leachate. With some progress through the phase, some degradation of cellulose begins, CO_2 concentration begins to decrease, and hydrogen is consumed with free levels reducing to zero. The low redox potential allows the methanogenic bacteria to become established with methane production occurring for the first time.

Phase IV. Assuming that an excess of VFAs has not inhibited development of methanogenic conditions, the methane fermentation phase or accelerated methane production now proceeds as methanogens become more dominant. The phase is characterized by steady methane production at 50 – 70 % of total gas production (the balance comprising mainly CO_2). Reduced VFA production and conversion of VFAs to methane and CO_2 results in a rise in leachate pH to about neutral values. Leachate BOD and COD concentrations, as well as conductivity, also decrease. Ammonia, released from protein reduction but not converted, is a conservative parameter for leachate analysis.

Phase V. This is the maturation phase which occurs after the readily available biodegradable matter in MSW has been degraded. The rate of gas generation diminishes significantly due to the removal of essential nutrients from leachate and the fact that substrates remaining in the landfill are very slowly degradable. Small amounts of nitrogen and oxygen may also be found in landfill gas depending on the landfill closure strategy. Leachate in this phase is characterized by the presence of humic and fulvic acids. Ammonia continues to be a conservative leachate parameter.

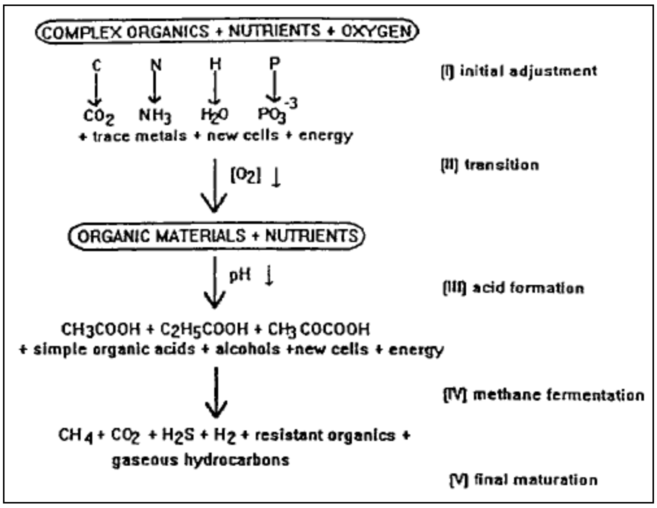


Figure 2.11: General scheme of decomposition process in a landfill (Saint-Fort, 1992).

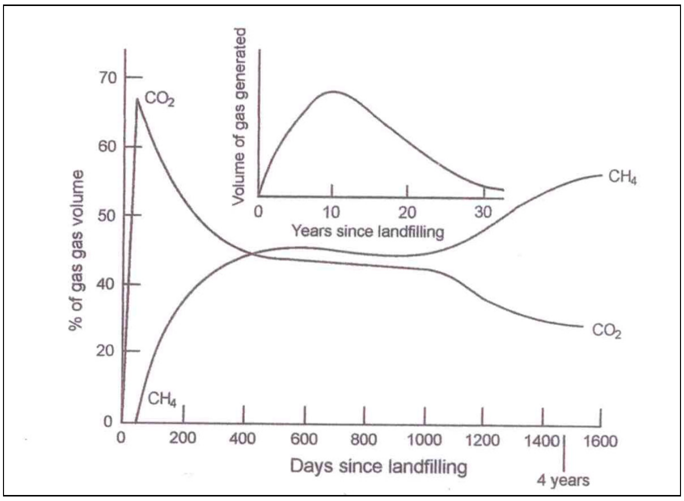


Figure 2.12: Generation of carbon dioxide, CO_2 and methane, CH_4 in landfilled waste. Variation of relative gas composition by volume with time after landfilling. Insert: Variation of gas quantity generated with time (Blight, 2010).

Three main groups of micro-organisms involved in anaerobic degradation of MSW

1. Fermentative

These are a large heterogeneous group of bacteria, also known as acidogens, which perform hydrolysis and organic fermentation. They comprise aerobes and both strict and facultative anaerobes and make up about 90 % of the total landfill microbial community.

2. Acetogenic bacteria

One set of this heterogeneous group of bacteria convert the products derived from fermentation into acetate, hydrogen and CO_2. There are extremely sensitive to hydrogen partial pressures. Another set, known as homacetogens, are a group of strict anaerobes which consume hydrogen and thus play an important role in regulating its partial pressure.

3. Methanogenic bacteria

These comprise two main groups, both of which are strict anaerobes. They are extremely substrate specific and dependant on other classes of bacteria to synthesize these since they cannot ferment carbohydrates. They require a low redox potential and sensitive to the pH of their environment (which should be neutral) and presence of oxygen.

It has been stated previously that refuse degradation is the function of physical, chemical and biological properties. However, it is not sufficient to understand refuse degradation solely based on the knowledge of the properties mentioned above. The following sub-sections will emphasize some of the factors influencing degradation of refuse.

Factors influencing waste decomposition

Several factors affecting or influencing waste decomposition in a sanitary landfill are well-documented and have been investigated by many researchers (e.g. Farquhar and Rovers 1973; Pohland, 1986; Rees, 1980; Christensen and Kjeldsen, 1989; Ehrig, 1989). A summary of various factors influencing refuse decomposition provided in table 2.7 below is followed by detailed discussion of such factors.

Influencing factors	Criteria / Comments	References
Moisture	Optimum moisture content : 60% and above (by wet mass)	Pohland (1986) ; Rees (1980)
Oxygen	Optimum redox potential for methanogenesis: -200mV -300mV below -100mV	 Farquhar and Rovers (1973) Christensen and Kjelden (1989) Pohland (1980)
pH	Optimum pH for methanogenesis: 6 to 8 6.4 to 7.2	 Ehrig(1983)/ Farquhar and Rovers(1973)
Alkalinity	Optimum alkalinity for methanogenesis : 2000mg/l Maximum organic acids concentration for methanogenesis : 3000mg/l Maximum acetic acid/alkalinity ratio for methanogenesis : 0.8	Farquhar and Rovers (1973) Farquhar and Rovers (1973) Ehrig (1983)
Temperature	Optimum temperature for methanogenesis : 40° 41° 34—38°C	 Rees (1980) Hartz et al. (1982) Mata-Alvarez et al. (1986)
Hydrogen	Partial hydrogen pressure for acetogenesis: below 10^{-6} atm.	Barlaz et al. (1987)
Nutrients	Generally adequate in most landfill except local systems due to heterogeneity	Christensen and Kjelden (1989)
Sulphate	Increase in sulphate decreases methanogenesis	Christensen and Kjelden 1989)
Inhibitors	Cation concentrations producing moderate inhibition (mg/ l) : Sodium 3500-5500 Potassium 2500-4500 Calcium 2500-4500 Magnesium 1000-1500 Ammonium(total) 1500-3000 Heavy metals : No significant influence Organic compounds : Inhibitory only in significant amount	McCarty and McKinney (1961) Ehrig(1983) Christensen and Kjelden(1989)

Table 2.7: Factors affecting degradation of refuse in a landfill (After, Yuen, 2001).

Moisture content and distribution

Moisture is fundamental for all microbial activity with a landfill ecosystem. Many investigations have shown that methane production rate increases with moisture content and is an accepted aspect of landfill management. Gas production increases exponentially up to an optimal moisture value of 60 % by wet mass and ceases completely below 10 %. In addition to its direct effect on microbial activity, moisture movement in landfills facilitates substrate, nutrient and buffer exchange, dilution of inhibitors and the distribution of bacteria. Free moisture movement can also affect the internal temperature of a landfill.

Source of energy

Two forms of energy are externally available to micro-organisms, light and exothermic reactions, and these are utilized by phototropic and chemotropic organisms respectively. Since available sunlight plays no significant role in a landfill interior environment, the former group is excluded and energy within a landfill is therefore limited to that derived from chemical reactions, possibly augmented by solar heat penetrating from the landfill surface.

Temperature

Three groups of methanogenic bacteria operate at different temperature ranges within a landfill. These are the pychrophilic, mesophilic and thermorphilic groups which operate at temperatures of less than 20 ^{0}C, 20 - 44 ^{0}C and over 40 ^{0}C respectively. The mesophilic group is by far the most responsible for methanogenesis within a landfill. Lysimeter studies show that methanogenic activity increases significantly (up to two orders of magnitude) with temperature increases from 20 ^{0}C to 40 ^{0}C with optimum temperatures reported in the range of 20-40 ^{0}C. It has been reported that methane production is sensitive to abrupt changes in temperature, even if the change is only a couple of degrees. The regulation of internal temperature of a landfill is twofold. Firstly, the internal temperature of a landfill can be readily regulated by the level of microbial activity. Secondly, the internal temperature is regulated by the degree of insulation provided by the media and cover material.

Oxygen

The presence of oxygen within the landfill ecosystem may inhibit activity of anaerobic methanogens. Research shows that optimal oxidation reduction potentials (ORP), or redox potentials (Eh), of between -200 mV and -300mV is essential for methanogenesis to function at optimum. Atmospheric oxygen ingress into the uppermost layers of a landfill can result in aerobic degradation there. Consumption of this oxygen generally limits this aerobic activity to be confined into the upper 1 m layer or so of the waste mass.

pH

This has a significant influence on methane production since methanogenic bacteria are only active within a narrow pH range of 6.5 – 8.

Alkalinity

Alkalinity is expressed as the concentration of calcium carbonate, this acts as an effective pH buffer and may significantly improve degradation efficiency by maintaining a close to neutral pH range in the landfill ecosystem. Literature surveys show that optimal alkalinity of 2000 mg/l is essential for methanogens, with a maximum allowable organic acids concentration of 3000 mg/l.

Nutrients

Nitrogen and phosphorus have been reported to be the main nutrients in a landfill ecosystem since biodegradable waste fractions exist as a carbon source. An optimal carbon to nitrogen ration of 16:1 has been suggested. Inorganic nitrogen generally occurs in landfills as ammonia, nitrate or nitrite, all of which play a role in the nitrogen cycle. Phosphorus is often the limiting macro-nutrient in landfills. A nitrogen to phosphorus ratio of at least 2:1 is required. Micro-nutrients include sulphur, calcium, potassium, iron, zinc, copper, cobalt and magnesium. Anaerobic processes are not nutrient-hungry and there are generally adequate supplies in a landfill, although nutrient-deficient pockets may exist.

Hydrogen

Hydrogen is both a substrate and product of anaerobic landfill processes. A low partial pressure of hydrogen during fermentation yields hydrogen, CO_2 and acetate while high partial pressure yields longer chains of VFAs and alcohols. During acetogenesis, if the partial pressure of hydrogen is too high, conversion of VFAS to acetate is impossible, often resulting in their accumulation and a subsequent drop in pH level. A hydrogen partial pressure below 10^{-6} atm has been reported as a requirement for acetogenesis.

Sulphate

It has been reported that landfill methane production will be reduced if the sulphate is present as sulphate reducing bacteria also consume hydrogen and acetic acid.

Inhibitors (Toxicity)

A number of substances have inhibitory or toxic potential which may hinder the growth of methanogenic bacteria. High cation and heavy metal concentrations, for example, inhibit methane production. Other substances such as aromatics, detergents, chlorinated hydrocarbons and phenolic compounds are toxic for the bacteria to flourish, although concentrations of these substances in a typical landfill are generally low unless co-disposal of industrial waste waters is being practised.

Other factors

Operational & process-based factors

A number of measures are available to landfill operators in order to exert some control over the factors identified above and so enhance waste degradation and stabilization.

The physical state of waste

A number of measures are available in this regards including the control and selection of waste with regard to its composition and moisture content, waste sorting and/or pre-treatment, co-disposal with wastes of high levels of inhibitory substances, the amount of waste in place (landfill size), shredding of waste and placement and compaction strategies, since these affect in-situ density.

Additives

A number of degradation enhancing additives can be applied to landfills. These include the addition of buffers, sewage sludge, ash, pre composted MSW or enzymes.

Infiltration and leachate recirculation

The amount of moisture available within a landfill is highly affected by the allowed rate of infiltration through the landfill cover. Where leachate collection and recycling systems exist and leachate recirculation is practiced, MSW degradation is aided by the provision of moisture, encouragement of water flux for microbial, substrate and nutrient transfer and dilution of inhibitors. As a result, lysimeters used in the present study were operated as bioreactors by means of recirculating leachate in order to enhance waste decomposition. Careful operation to prevent overloading the in-situ leachate treatment capacity of the landfill is necessary.

CHAPTER THREE: MATERIALS AND METHODS

3.1 EXPERIMENTAL SET-UPS

3.1.1 Laboratory Lysimeters

The use of laboratory lysimeters has gained wide acceptance as a means to simulate actual landfill conditions. In simple terms, “lysimeter” is a term used to describe model or laboratory simulated landfills. A lysimeter can also be a full scale experimental section on a landfill.

Owing to the previous success in the use of lysimeters by many researchers, the present study has followed the well-tried paths paved by earlier researchers. To mention a few, Qasim (1965) successfully used concrete cylinders with different heights to study the emissions from refuse. Subsequently, material types for lysimeter construction have evolved from not only using concrete, to a wide range of materials such as glass, stainless steel and various types of plastics.

The use of lysimeters to understand landfill processes offers a wide range of advantages; for this reason, its use cannot be underrated. The sizes of lysimeters which have been used in the past ranged from 2 ℓ to 6.4 m^3 according to Barlaz *et al.* (1992) and Pohland (1980). Smaller lysimeters can be located in a laboratory, and making for a more cost effective approach, affordable, rapid provision of results, and low cost implications etc. As a result, this may enable one to conduct multiple tests that cannot be readily economic feasible with pilot-scale landfills, and this justify its use in the present study.

3.1.2 Lysimeter construction and design

Lysimeters aimed at producing similar conditions to those of an actual landfill require selecting the best material type for construction. The vast experience in the use of laboratory lysimeters to simulate landfill processes shows that some of the material being used for lysimeter construction had the tendency to influence the landfill process more than would have been expected from a prototype landfill. Therefore, it should be a fundamental pre-requisite when opting for the material type for lysimeter construction that it should not by any chance permit negative impacts on the waste contained within the lysimeter. This is in substantial agreement with the studies by Shamrock, 1998;

Morris, 2001 and Rohrs, 2002, who used 50 litre HDPE drums for lysimeter construction. The principal factor which dictated the use of HDPE drums over the variety of materials such as metals and other material types was benchmarked by the ability of HDPE drums to resist the chemical aggressive conditions of degrading refuse.

It has been already mentioned that for a lysimeter aimed at producing similar conditions to that of the actual landfill the best material type for construction must be selected; therefore, the present study made use of sections of UPVC water pipe because of UPVC's ability to resist the chemical aggressive of degrading refuse (Moola, 2001). A study by Rohrs (2002) singled out the four main components which are required of laboratory simulated landfills or lysimeters *viz.,* a sealed container, an irrigation system, a collection system and a gas monitoring or outflow port. The present study has incorporated all these essential components, with slight modifications.

The bottom part of each lysimeter was equipped with a T-piece to serve two purposes: 1) Both ends of the T-piece were equipped with valves, one valve was used as a leachate sampling port. 2). the other valve was used to monitor any gas likely to accumulate at the bottom of the lysimeter, and included a water manometer to measure the pressure of the gas and, at the same time, to contain the gas by means of the water column. Prior to filling the lysimeters with waste of a particular composition, the bottom part of each lysimeter was provided with layer of quartz sand for leachate filtration and drainage. Subsequently, individual lysimeters were filled by hand and compacted in layers using a Marshall Compaction Hammer (see the sub-section on lysimeter filling). A headspace of 30 mm was left in all of the lysimeters. A gas vent incorporating a sampling port was installed in the side of the lysimeters at the headspace level. Another port was installed at the mid-height of each lysimeter for gas monitoring. This incorporated a plastic tube with a water trap to prevent gas escaping from the lysimeter and to measure the gas pressure. An irrigation system in the form a of valve attached to a nylon plate positioned at the top of each lysimeter for water additions without allowing the entry of air. The plates positioned on top of individual lysimeters were tightened against two metal end-plates positioned at the bottom of each lysimeter by means of threaded mild steel rods.

All the ports were sealed and made air tight by making use of LOCKTITE 111, epoxy, tape and silicon sealer. Each lysimeter was double wrapped in bubble wrap as insulation to control internal temperature.

A total of 6 lysimeters used in this study; each with length of 63 cm, and internal diameter of 20 cm, were kept in a small windowless room Figure 3.1 shows the various components of a laboratory lysimeter. Figure 3.2 shows a layout of 6 lysimeters facing fan-heaters to heat the room and maintain a constant temperature.

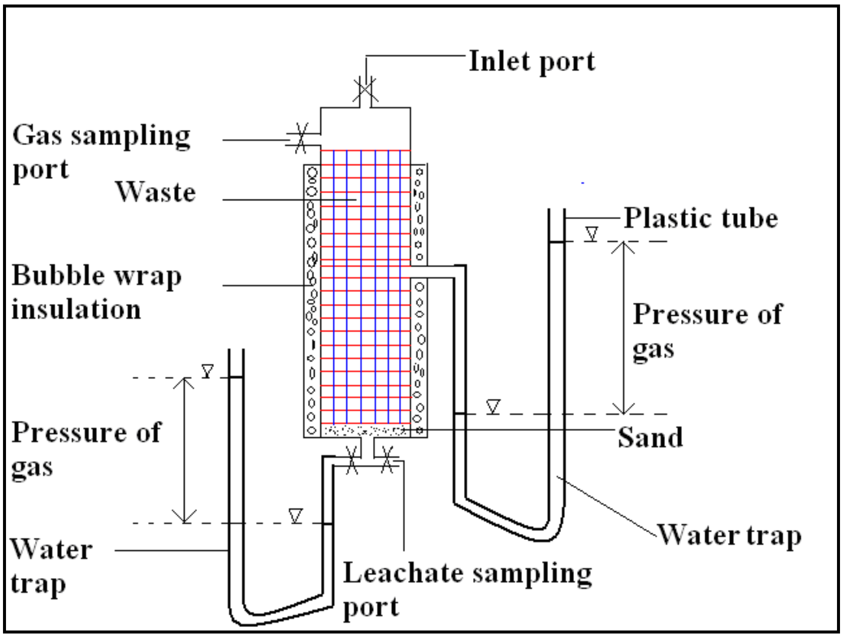


Figure 3.1: Components of the laboratory lysimeter (not drawn to scale).

Figure 3.2: Layout of fan-heaters and bubble-wrapped lysimeters.

3.1.3 Lysimeter properties

Each of the six identical lysimeters was loaded with waste of different compositions to represent either “poor”, “rich” or mixed “poor” and “rich” waste. The 6 lysimeters were divided into two sets, namely sets A and B. Each set comprised of three lysimeters. Set A lysimeters were named AL1, AL2 and AL3, while set B lysimeters were named BL1, BL2 and BL3. The first letter for each lysimeter in a particular set denotes a set number, followed by lysimeter (L) number in that particular set. It is important to note that corresponding lysimeters of each set, e.g. AL1 and BL1 were filled with same waste mix, but at different moisture contents.

There are four key issues to be considered before one can embark on the use of laboratory lysimeters. These include the type of waste to be used; the waste mix; the amount of waste needed, and lastly, the density required. It has been mentioned that the lysimeters used in this study were 63 cm long with an internal diameter of 20 cm. Based on this information; the following steps were followed to compute the amount of waste needed:

$$\text{Area of a circle: } A = \pi D/4 = 0.0314 \text{ m}^2$$

$$\begin{aligned}\text{Volume of a cylinder: } V &= Ah \\ &= 0.0314 \text{ m}^2 \times 0.63 \text{ m} \\ &= 0.0198 \text{ m}^3\end{aligned}$$

$$\text{Density: } \rho = \frac{M}{V}$$

$$\therefore M = \rho V$$
$$= 800 \times 0.0198$$
$$= 16 \text{ kg (dry waste)}$$

However, the waste received in a typical landfill is deposited at a range of moisture contents, but the mass of waste calculated above is on a dry mass basis and compaction cannot be readily achieved on dry waste. Therefore, the mass of dry waste was re-calculated using field capacity water contents for each waste mix.

A study by Rees (1980) revealed that the methanogenic phase can be attained if waste in a lysimeter is compacted within a range of 500 to 1000 kg/m^3 wet density. Therefore, 800 kg/m^3 density was adopted as the target in the current study in line with the research findings by Rees. However, 800 kg/m^3 density was not attained in all the lysimeters, especially in set B lysimeters which were compacted at their half field capacity (Tables 3.1 and 3.2 for waste mass and densities).

3.1.4 Waste characterization

(a) Waste composition

The present study compares leachate generation potential between B^+ and B^- landfills and decomposition of MSW under these conditions by means of using laboratory lysimeters. To date, South Africa has a number of landfills operating under B^+ and B^- sites, and depending on their respective locations, either poor or rich waste or mixed poor and rich waste is disposed of in those landfills. In this study, basic components of synthetic MSW resembling waste from ''poor'' and ''rich'' suburbs of South Africa were used. Three waste components were used. These were paper, grass clippings and fly ash. Papers were sourced from the offices in the School of Civil and Environmental Engineering at the University of the Witwatersrand in Johannesburg, and included mixed proportions of computer papers and newspapers. About 8 bags of grass from the residential area around Johannesburg were kindly provided by one of the staff members and others were sourced from Wits University lawns.

Owing to the high moisture content of the grass on arrival, it was spread on the laboratory floor, as well as on top of flat concrete bench tops for drying. The ash component could not be obtained even from large waste management companies serving the metropolitan city of Johannesburg. For this reason, fly ash from Lethabo Power Station was used to represent the ash fraction and was kindly provided by Ash Resources Ltd. A big advantage of the chosen components was that they were reasonably uniform in composition, much more uniform than if real municipal solid waste components had been used.

The waste used in the present studies was intended to mimic waste of poor and rich suburbs in terms of paper, ash and putrescible contents. The guideline used to decide whether waste is “poor” or “rich” was based on a comparative study of the decomposition processes and products of rich and poor refuse in South Africa undertaken by Shamrock (1998). The poor waste is typically characterized by a high ash content, whereas rich waste has a low ash content.

Set A lysimeter: Waste components			
Lysimeter number	Waste type	Mass (kg) (Dry)	(%) of total waste
AL1	1. Paper	1.788	30
(Poor)	2. Grass	0.596	10
	3. Ash	3.576	60
5.960 kg			
AL2	1. Paper	1.868	40
(Poor/Rich)	2. Grass	0.934	20
	3. Ash	1.868	40
4.670 kg			
AL3	1. Paper	1.905	50
(Rich)	2. Grass	1.143	30
	3. Ash	0.762	20
3.810 kg			

Table 3.1: Set A lysimeter waste components.

Set B lysimeter: Waste components			
Lysimeter number	Waste type	Mass (kg) (Dry)	(%) of total waste
BL1	1. Paper	2.607	30

(Poor)	2. Grass	0.869	10
	3. Ash	5.214	60
BL2	1. Paper	2.916	40
(Poor/Rich)	2. Grass	1.458	20
	3. Ash	2.916	40
BL3	1. Paper	3.08	50
(Rich)	2. Grass	1.848	30
	3. Ash	1.232	20

Table 3.2: Set B Lysimeter waste components.

The above tables show the three dry waste components and their mass proportions. Lysimeter 1 waste component in both sets is characterized by similar high proportions (60 %) of ash content and represents waste from poor communities. On the other hand, lysimeter 3 in set A and B are both characterized by relatively low ash fraction, and for this reason represents waste coming from rich suburbs. Lastly, lysimeter 2 in both sets shows a balance between poor and rich waste and it is referred to as 50: 50 poor and rich waste.

As has been stated previously, 16 kg of dry waste was required to fill up the lysimeters, which was computed based on the lysimeter dimensions as well as the density required. However, the literature survey shows that waste (MSW) generated around the world arrives at landfill sites with varying moisture contents. Therefore, it was important to adjust the water content prior to filling the lysimeters. Among many physical properties of waste, water content and field storage capacity were determined in order to provide water content of waste having a such as that in a typical landfill (see the following sub-section on the physical characterization of waste). The field storage capacity data was used to calculate the amount of waste required, as well as the amount of water for waste of a particular mix. All the calculations were validated to ensure that the amount of waste required and water when mixed together added up to 16 kg (now of waste + water).

The mass of solids (waste), M_s, is defined by the following equation:

$$M_s = (M_w + M_s) \times \frac{1}{1 + FC} \qquad \text{Eqn 3.1}$$

or $(M_w + M_s) = M_s (1 + FC)$

In equation 3.1 $(M_w + M_s)$ and FC are the masses of (water + solids) and field storage capacity, respectively. The mass of water, M_w, was calculated by using the following equation:

$M_w = FC \times M_s$ Eqn 3.2

AL1: $M_s = 5.96$ $FC = 1.68$

$M_w = 5.96 \times 1.68 = 10.01$

$M_s + M_w = 5.96 + 10.01 = 15.97$

Where: FC and M_s are the field storage capacity and mass of solids (waste) respectively. In both equations 3.1 and 3.2, the relationship between mass of solids and field storage capacity is inversely proportional. This is in substantial agreement with the results of waste masses for set A and B lysimeters after the field storage capacity data had been used to compute the mass of waste required. Table 3.3 & 3.4 shows a decrease in mass of waste as the field storage capacity increases. (See page 65 for determination of FC.)

Lysimeter numbers	Mass of dry solids (waste) in kg	Mass of water (kg)	Mass of solids + Mass of water (kg)	Field storage capacity on dry mass basis (%)	Initial moisture content on dry mass basis (%)	Relative values of FC
AL1	5.96	10.01	15.97	168 %	1.43 %	1
AL2	4.67	11.21	16	240 %		1.43
AL3	3.81	12.04	15.97	316 %	1.50	1.88

Table 3.3: This table shows mass of water used for waste mixture and field storage capacity of waste in Set A lysimeter components, as well as the initial moisture content of waste.

Lysimeter numbers	Mass of solids (waste) in kg	Mass of water (kg)	Mass of solids + Mass of water (kg)	Half storage capacity on dry mass basis (%)	Moisture content in dry mass basis (%)

BL1	8.69	7.31	16	84 %	1.43 %
BL2	7.29	8.69	15.98	120 %	
BL3	6.16	8.84	15	158 %	1.50 %

Table 3.4: This table shows mass of water used during waste mixture and field storage capacity of waste in Set B lysimeter components, as well as the initial moisture content of waste.

Determination of moisture content and field storage capacities of waste

Specimens of waste were prepared to represent the waste mix for individual lysimeters. Specimens of refuse (1 kg each) were placed into a 25 litre bucket; half filled with water and thoroughly mixed with a shovel. Each specimen of waste mix for a particular lysimeter was placed on a sieve that rested on planks positioned over a pan (figure 3.3) below. A shovel was used to place the waste on the sieve, and the sample was inundated with the water that remained in the bucket after mixing and allowed to drain for 24 hours (figure 3.4). The top of the sieve was covered with a plastic in order to avoid moisture loss. After 24 hours, a representative sample was taken and weighed. The gravimetric water content was then obtained by oven drying at 50^0 C until a constant mass was reached. Results of moisture content and field storage capacities are reported in Table 3.3 and 3.4.

Figure 3.3: Sieve filled with waste.

Figure 3.4: Water accumulated in a gathering pan after 24 hrs.

3.1.5 Filling the lysimeters

Prior to filling the lysimeters with waste of a particular composition, a shredder was used to reduce the size of the pieces of waste in order to allow for easier mixing and homogenizing and to simplify the filling of the lysimeters (figure 3.7). Shredded wastes contained in plastic bags were transferred to a concrete mixer for final mixing (figure 3.5). Because of the generation of dust caused by the ash during mixing, the mouth of the concrete mixer was covered with plastic to avoid liberation of the dust. Water was also used to reduce the dust, as well as to promote rapid mixing of waste. When a uniform mixture was attained the waste was tipped into a pan. This was followed by filling up the lysimeters in roughly four layers by hand and compaction was achieved using a Marshall compaction hammer (figure 3.6). Six lysimeters were filled up in a period of a week.

Figure 3.5: Mixing of the refuse using a concrete mixer.

Figure 3.6: Marshall Compaction hammer resting on compacted waste.

Figure 3.7: Trim Rite Garden Shredder Machine.

3.1.6 Controlled parameters

There are several controlled parameters when one deals with laboratory lysimeters and these parameters are difficult to control in a typical landfill environment. In the case of this study, water addition and temperature were two controlled parameters since a landfill is subjected to both precipitation and changing temperatures.

(a) Water addition

Water contributes most to a higher degree of decomposition and leachate generation. In this study, leachate was recycled in the lysimeters that were at field capacity (set A lysimeters) after a period of two months. After a representative sample had been taken for leachate analysis, the remaining leachate drained from the lysimeter was recirculated together with distilled water to compensate for the volume of leachate which had been used for analysis. The leachate was recycled back into the lysimeters in order to maintain a constant water content, and leachate was recycled on a weekly basis. The amount of water added was always equal to the volume of leachate extracted from each lysimeter. The screw-operated ram of a pore water pressure measuring apparatus was used to inject a measured amount of water into the lysimeters without allowing air to enter, or gas to escape as depicted in both figure 3.8 and 3.9.

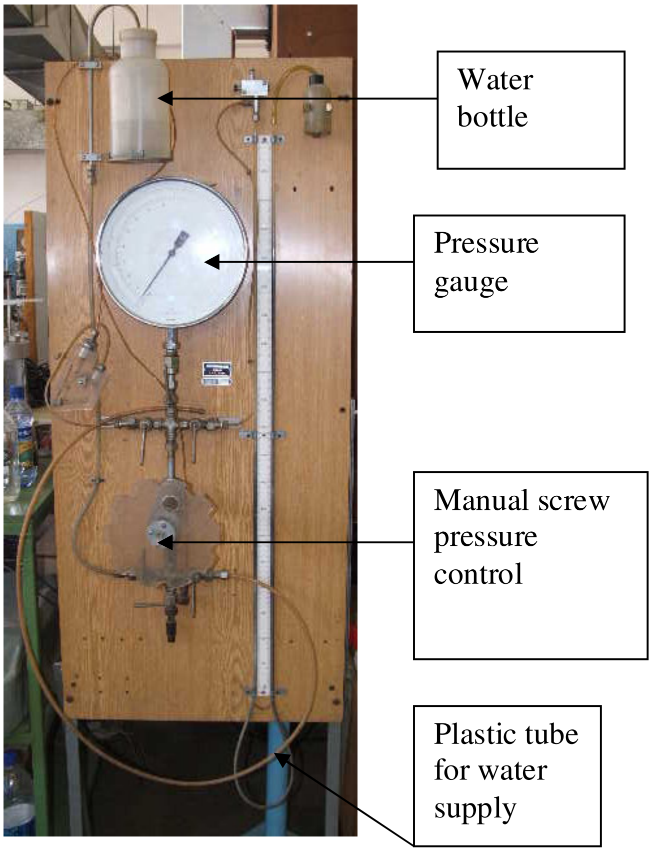


Figure 3.8: Manual Pore Water Pressure Apparatus.

Figure 3.9 shows a plastic tube supplying water into the lysimeter.

(b) Temperature

Landfilled wastes are characterized by the presence of various micro-organisms. These micro-organisms, especially methanogens, have specific temperature ranges under which they function optimally, while at higher or lower temperatures they die or go dormant. Different temperature ranges are reported in literature by various researchers as the optimal temperature for methanogens (Rees, 1980; Hartz et al. 1982; Mata-Alvarez et al., 1986). However, a range of temperatures from 30 to 40 ^{0}C have been used by various researchers as the optimal temperature for methanogens (Rohrs, 2002).

The 6 lysimeters used in this study were wrapped with bubble wrap a week after the lysimeter were filled. As shown in figure 3.2, two to three heater-fans were used to maintain an external steady temperature in lysimeters room. The main advantage of this type of heater is the ability to control the temperature by means of oscillation of the fans and by setting a required temperature range. A thermometer suspended by a string was placed close to the lysimeters to measure the air temperature.

3.1.6 Monitored parameters

In this study, it was intended to study three parameters: temperature, leachate volume and gas measurements.

(a) Temperature

Unfortunately, temperature was not recorded from day one of the experiment; however, the available data for 20 days during test period is plotted in the graph below. It is also important to note that the plotted points fall within the previously reported temperature ranges for methanogens by various researchers.

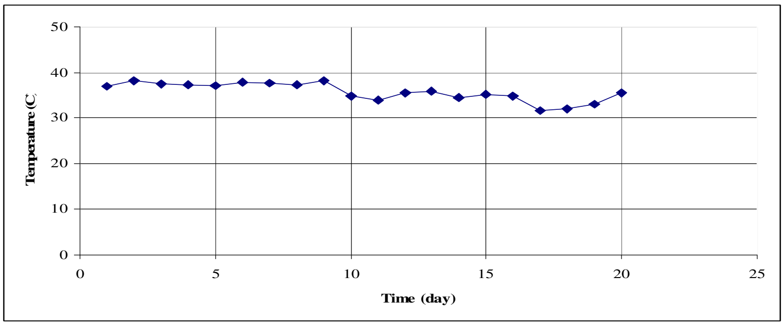


Figure 3.10: Temperature values over the period of monitoring.

(b) Leachate volume

It has been stated previously that the 6 lysimeters used in the present study were placed in the laboratory during mid July 2010 and all were left for two months before recycling of leachate was commenced. However, generation of leachate was noticed in the lysimeters at field capacity after a period of two weeks. Leachate sampling was undertaken twice in the period of two months, and thereafter, sampling was done weekly when possible (Appendix B).

(c) Gas measurements

No gas analysis was performed in the present study due to technical problems with the GC chromatograph which was non-functional. A replacement column was ordered, but arrived too late for use in the project. However, gas pressure was measured in the tube for monitoring gas pressure.

3.1.7 Sampling and Analytical techniques/methods

Leachate

Leachate sampling was conducted on a weekly basis in most cases. The principal reason for the regular sampling was to assess any trend likely to appear in leachate production with respect to time. Set A lysimeters were left unirrigated for the last three months of monitoring, and thereafter, samples of leachate were collected during the last day of monitoring, day 262. A sample was collected by opening a tap equipped with a plastic tube at the bottom of a lysimeter leading to a leachate collection bottle. Once a tube was full, the tap was shut and leachate contained within the tube was transferred into a sampling bottle into to obtain a representative sample for analysis. This process was repeated every time the tube was full of leachate. After a representative sample had been sent to the laboratory, the remaining amount of leachate was recirculated back into the respective lysimeters in order to maintain the moisture equilibrium within the lysimeters.

The samples of leachate collected from the individual lysimeters were analysed for several parameters in the civil engineering department water quality laboratory, University of the Witwatersrand. pH and conductivity were measured using JENWAY pH and conductivity meters. In addition, COD, PO_4, NH_4^+ and Cl^- were determined by using a Merck SQ118 spectrophotometer. Samples were diluted to the appropriate measuring range and then chemical reagents were added in accordingly. Some parameters such ammonia and chloride were analysed immediately after sampling, while others were analysed within the same week of leachate sampling.

CHAPTER FOUR: QUANTITY AND QUALITY OF LEACHATE GENERATED FROM THE LABORATORY LYSIMETERS – ANALYSIS, INTERPRETATION AND DISCUSSION OF RESULTS

4.1 INTRODUCTION

The introductory chapter (Chapter One, section 1.2 – Problem Statement) stated the nature of the problem that needed to be addressed and showed that there was a need for the current study to close the identified gap in knowledge. It was important to test the validity of the hypothesis formulated for the study (See chapter one, section 1.5). Through the literature survey, it had been noted that the primary objective of the present study can be achieved by creating in the laboratory a soundly engineered model representing typical landfill conditions (Details for the experimental set-up are provided in chapter three: Materials and Methods).

The prime reason that motivated the present study was the need to investigate the potential for leachate generation and the process of decomposition of MSW through laboratory simulated landfills (i.e. for landfills situated at B^+ and B^- sites). This chapter deals specifically with the quantity and quality of leachate generated in different lysimeters, i.e. different in terms of having similar waste types, but with varying proportions, as well as differing in their field storage capacities and initial moisture contents. The quantity of leachate generated is presented in terms of water balance measurements recorded of the individual lysimeters.

This chapter will present the results obtained during the experiment to aid addressing the above-mentioned issues, in a scientific defensible way. More specifically, the findings from this chapter will allow one to assess the extent to which wastes having different proportions of paper, ash and putrescible content may affect leachate generation in terms of quantity, as well as quality. In this chapter, successive sections present the water balance measurements for lysimeters sealed at their field capacities (i.e. set A lysimeters) and results for leachate quality parameters such as pH, conductivity, COD, ammonia, chloride and phosphate are also presented.

4.2 LEACHATE GENERATION IN THE VARIOUS LYSIMETERS

Leachate generation from landfills has been predicted by means of the water balance method for more than three decades. Since the inception of this method, a range of predictive models has been developed in line with the water balance method in order to enhance the knowledge of water behaviour in a typical landfill (the water balance method is presented in detail in chapter two-section 2.3.2). Most of these models have been successfully used for prediction of landfill leachate production around the world. However, the precision and accuracy of the models do not portray the actual behaviour of water within a landfill, but only relate the outputs to the corresponding inputs. Nevertheless, the water balance method remains a baseline for predictions of leachate generation in sanitary landfills.

The law of conservation of mass predicts that the volume of water entering the waste body must equal to the amount of water exiting at its base as leachate minus the water stored within the waste. As a result, equation 2.2 (L = P – ET) previously reported in chapter two, section 2.3.2, can be further simplified, for application to the lysimeters, to:

$$L = P \qquad \text{Eqn 4.1}$$

In this case, L is taken as the amount of leachate generated (in ml), while P denotes the volume of water added (in ml) in the lysimeters. The evapotranspiration (ET in Eqn 2.2) which is an integral part of the climatic water balance equation in the Minimum Requirements has been omitted due to the fact that all the lysimeters used in the present study were sealed and not exposed to evaporation regime. By so doing, it has allowed conservative measurement of leachate generation as well as for examining the relationship that exists between water input and output in the lysimeters based on the assumption made above.

4.2.1 Details of leachate generation

After two weeks of continuous monitoring when all the lysimeters had been filled and sealed, two of the three lysimeters filled at field storage capacity (i.e. AL2 and AL3) started to generate significant volumes of leachate, whereas AL1 did not generate leachate until day 66. No leachate generation was detected in the B lysimeters filled at half field capacity. The early production of leachate from set A lysimeters is attributable to the relatively high moisture content that characterizes wastes at their field capacities, as well as the compaction density of the waste.

It was decided to leave all the lysimeters unirrigated for the period of two months in order to assess the potential for leachate generation in set B lysimeters. However, there was no sign of leachate generation even two months after the waste had been placed in set B lysimeters. Hence, set B lysimeters were left unwatered for the rest of the test since there was no leachate to be recycled. As a result, this section specifically provides analysis of results for leachate generation and water balance measurements recorded from set A lysimeters. At the end of the programme, the B lysimeters were artificially leached by adding half of the water necessary to bring them up to their original field storage capacity. The leachate sampling ports at the base of B lysimeters were closed before irrigating them, and were left closed for 7 days. On the 8th day, sealed sampling bottles were connected to the leachate sampling port and the drainage valves were opened. The quantity of leachate draining out of each lysimeter during the next 7 days was measured and the leachate was analysed, as for leachate from the A lysimeters.

Set A lysimeters

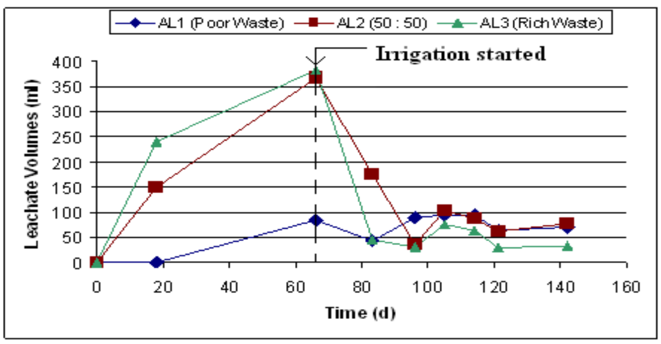


Figure 4.1: Leachate generation in set A lysimeters over the period of monitoring.

Figure 4.1 graphically presents measurements of leachate generation (y-axis) against the number of days during the period of monitoring (x-axis). It has been stated earlier that the lysimeters filled at field capacity started to generate appreciable amounts of leachate prior to water addition in the lysimeters. By day 18, about 150 and 238 ml of leachate had been generated in lysimeters AL2 and AL3 respectively, while there was no leachate generated in lysimeter AL1 (which is also one of the lysimeters prepared at field capacity). It was important to ascertain the possible factor/s which retarded the early generation of leachate in lysimeter AL1.

It is believed that the high proportion of ash (60 % by mass) characterizing lysimeter AL1 (poor waste), apart from being the most absorbent component of the waste, entirely filled the void spaces between the paper-grass waste mix (Details for waste mix for all lysimeters are presented in Table 3.1 and 3.3). This probably resulted in a higher density of the waste in lysimeter AL1 as compared to waste in lysimeters AL2 and AL3. For this reason, the storage of water in AL1 remained above that in AL2 and AL3.

The highest volumes of leachate in lysimeters AL2 and AL3 were produced by day 66 (i.e. two months after waste had been placed into the lysimeters) as shown in figure 4.1 above, with AL2 reaching 368 ml and AL3, 382 ml. Leachate emission from AL1 commenced on day 66 when 84 ml of leachate was collected. Figure 4.1 shows a large drop in leachate production both in AL2 and AL3 by day 83. Since the irrigation of all the lysimeters began on day 66, the leachate collection pattern depicted in figure 4.1 must have resulted from the irrigation day on 66, starting from day 83.

Initially, lysimeter AL3 (i.e. rich waste) was generating significant volumes of leachate followed by AL 2 (50: 50), while AL1 (Poor Waste) generated the least amount of leachate up to day 66. The generation of leachate up to day 66 can be explained in terms of the relationship between field capacity and ash content. It had been noticed that as the ash content increased (Table 3.1) the field capacity decreased. An alternative explanation is that the poor waste, having the lowest field capacity, started out with least water and therefore could be expected to produce least leachate (Table 3.3). From day 83 onwards all of the lysimeters reached an approximately steady state in terms of leachate generation, although lysimeter AL3 generated the smallest amount of leachate.

The cumulative leachate generated over the period of monitoring is shown in figure 4.2. Figure 4.2 shows that the amount of leachate generated in the lysimeters increased from poor to rich waste. There is a huge difference in leachate production in lysimeter AL1 as compared to leachate produced in both lysimeters AL2 and AL3. The difference of leachate generation in AL2 and AL3 as compared to AL1 by day 142, was about 340 ml (900 – 560) or 60 % more. Curve AL3 is much higher than curve AL1 and all curves showed a decline in the rate of leachate generation by day 142.

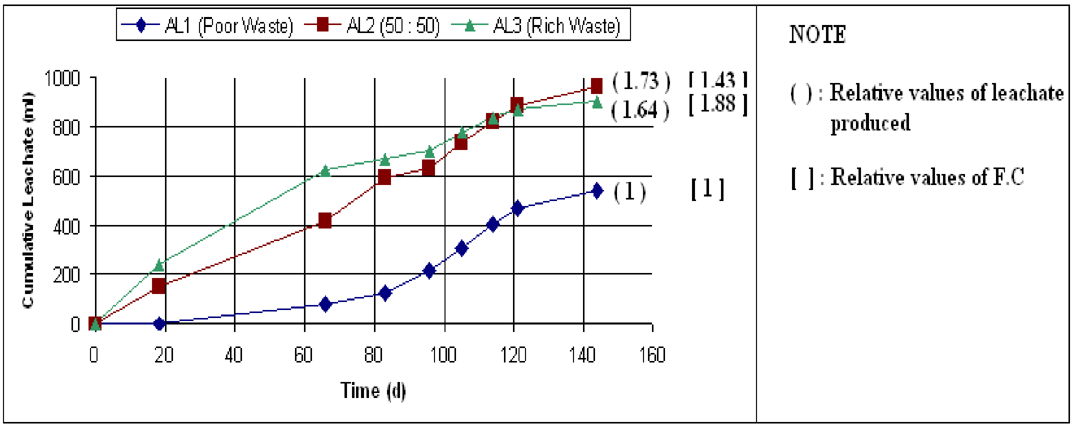


Figure 4.2: Cumulative leachate generation in set A lysimeters over the period of monitoring.

The figure above appears to provide evidence that leachate generation is affected by waste type as well as water volumes. Considering all these factors and others such as climate, it is apparent that two landfills one receiving poor waste and the other rich waste, may generate distinctly different volumes of leachate. In this case, the lysimeter filled with poor waste generated only about 60 % of the leachate generated by a lysimeter filled with 50 % poor and 50 % rich waste and with a lysimeter filled with rich waste. Although all three had been filled with waste at field capacity, they did not start the same mass of water because the field capacities differed. In terms of leachate collection and management systems a landfill disposing of poor waste may therefore be operated at lower cost than a landfill receiving rich waste. If landfills containing Poor, 50: 50 and Rich waste were to receive the same amount of rain, the order of quantity of leachate generated would probably be lowest: Rich waste; highest : Poor waste; in between : 50: 50.

4.2.2 Water balance considerations

The water balance results presented below embody both the controlled and monitored parameters of water addition and leachate generated. Each volume of leachate collected was immediately replaced into the lysimeters by an equal volume of water in order to maintain a constant water content in the waste. The difference (in percentage) between total water added and leachate generated was calculated for all the lysimeters, and a range of water additions was made (according to eqn 4.1) bracketing the climatic division from B^+ to B^- for each lysimeter i.e., in this experiment, from the lowest field capacity (B^+) to the highest (B^-).

Table 4.1 shows none of the lysimeters attained 100 % of leachate generation. The imbalances in water input and subsequent output is attributed to water lost during the process of waste decomposition and changing moisture storage capacity of the various wastes. For instance, lysimeter AL3 was characterized by a high field storage capacity than lysimeters AL1 and AL2.

Table 4.1: This table shows a range of water additions and total volumes of water added and leachate generated, as well as the percentage difference between total water added and leachate generated.

Lysimeter Numbers	Total water added (ml)	Total leachate generated (ml)	Leachate generation as % of water added
AL1	469.94	456.82	91
AL2	690.56	539.86	78
AL3	422.43	277.94	66

4.3 LEACHATE QUALITY: SET A LYIMETERS

One of the major aspects when dealing with landfilling of MSW is that of negative impacts to the environment. It is accepted that once water has percolated through landfilled waste, contamination of surface and groundwater by leachate will occur unless the leachate is collected and treated before being discharged into the environment. Numerous methods have been developed to aid in understanding the leachate quality; these methods determine the parameters affecting the quality of leachate, as well as factors in the waste and landfilling process that affect these parameters. Several parameters in the present study were measured, in line with the Minimum Requirements for Waste Disposal by Landfill in South Africa (1998). These were: pH, conductivity, COD, ammonia, chloride and phosphate, and the analysis and discussion of results for these parameters appear below. All the analyses were performed in the in-house laboratory in the School of Civil and Environmental Engineering, University of the Witwatersrand.

4.3.1 pH, CHEMICAL OXYGEN DEMAND (COD) AND CONDUCTIVITY

Among many parameters, COD has been successfully used to determine the quality of leachate. According to the American Public Health Association (APHA, 1989 quoted by Moola 2001), the COD value marks the oxygen level equivalent to the organic matter content in a sample that will easily undergo oxidation by a strong chemical oxidant. Rohrs (2002) defined COD as a measure of the amount of oxygen that is required to completely oxidize all the compounds in a solution. In simple terms, COD is used to measure the amount of organic matter in leachate although a third of contaminants in leachate is made up of inorganic compounds (Kylefors et al., 1999).

The COD results obtained in this study were compared with pH results in order to enhance the knowledge of waste decomposition process in the lysimeters. It is believed that as the organic compounds start to decompose, organic acids and alcohols are produced before methane can be formed and this often results in low pH values (Rohrs 2002). For this reason, pH has been widely used as an indicator to mark various phases that waste undergoes during the process of refuse degradation.

Figure 4.3 and 4.4 show a strong positive relationship between COD and pH values in set A lysimeters. Lysimeter AL1 (Poor Waste) was within the pH range of 7.50 – 9.05 for the entire period of the test, which is the most suitable pH range for optimum functioning of methanogens. A close look at figures 4.3 and 4.4 shows a decline in COD values as the pH level increases (this is more evident in lysimeter AL1). The higher pH level coupled with low COD values in lysimeter AL1 indicates that the methanogenic phase may have been reached first in lysimeter AL1 rather than in lysimeters AL2 and AL3. The early onset of the methanogenic phase is attributed to the high fraction of ash (60 % by mass) within lysimeter AL1. This implies that the fly ash might have acted as a buffer making conditions more conducive for the establishment of methanogens. However, different phases of degradation especially the methanogenic phase, which is often known as the phase for methane production, could have been easily defined or delineated in the lysimeters if only gas analysis had been possible in the present study. As a result, the criteria used for defining various stages of waste decomposition, more importantly the initial phase from methanogenic phase in this case, was restricted to changes in leachate quality parameters observed during the test. In lysimeter AL3, the pH level remained almost constant and below 6 during the time of monitoring. This shows that acid conditions prevailed throughout the experiment and as a result, some of conditions required for establishment of methanogens might not have existed or were inhibited by the acid conditions.

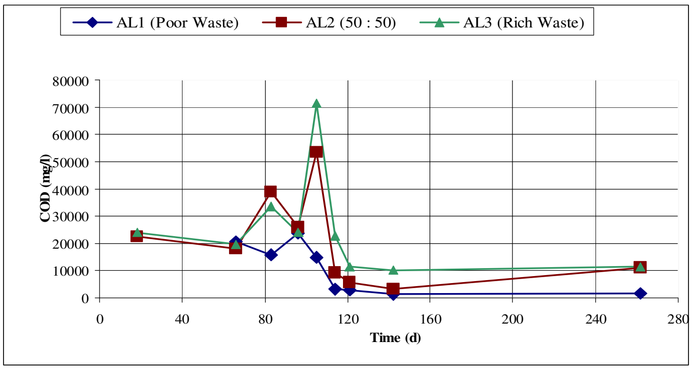


Fig 4.3

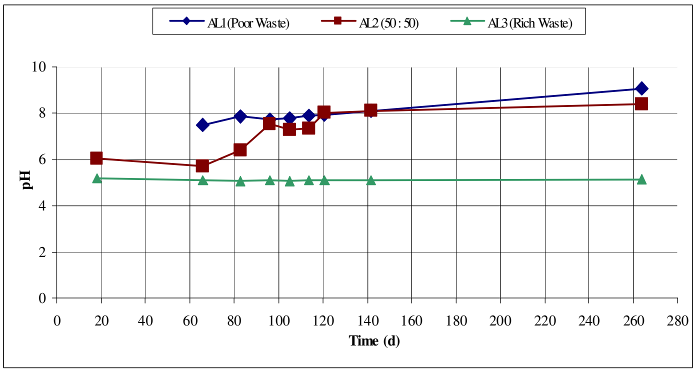


Fig 4.4

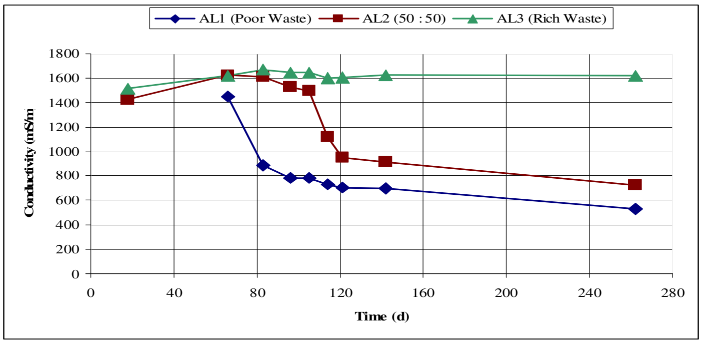


Fig 4.5

Figures (4.3, 4.4 and 4.5) shows respectively, COD, pH and conductivity values for set A lysimeters over the period of monitoring.

The relationship between pH and conductivity is inverse, as shown in the figures 4.4 and 4.5. AL1 exhibited the highest conductivity peak of 1400 mS/m at day 66, followed by a sharp decline in conductivity from day 66 to day 83. Thereafter, conductivity values started to decrease until a steady state was reached. Conductivity values in AL2 followed a similar pattern to that of AL3 until day 83, while AL3's conductivity remained relatively constant for the entire period of monitoring. The high conductivity values in AL3 may be the result of strong acidogenic conditions caused by dissolution of organics and inorganics.

4.3.2 Ammonia

The Department of Water Affairs and Forestry South Africa (DWAF, 1996) has specified a permissible concentration of ammonia in drinking water of 1mg/l. However, leachate is characterized by high concentrations of ammonia ranging from 30 – 3000 mg N/ℓ with an average of ~ 750 mg N/ℓ. The migration of leachate from a landfill site into surface and groundwater would therefore be detrimental to both the human health and aquatic species. Crawford and Smith (1995) postulated that ammonia is toxic and most lethal to fish at concentrations between 2.5 – 25 mg/ℓ.

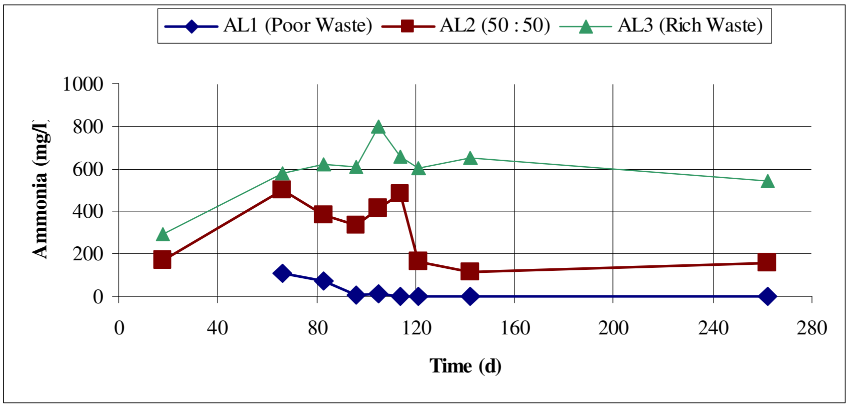


Figure 4.6: Ammonia concentrations for set A lysimeters over the period of monitoring.

Figure 4.6 above shows that ammonia loads were high in rich waste and much less in poor waste. The ammonia concentration in AL2 lies between that of poor (AL1) and rich waste (AL3). It is evident that ammonia concentration varies as waste grade from poor to rich. In AL3, the ammonia concentration remained high throughout the time of monitoring with a peak of 800 mg/l at 105 days, while AL1 showed an initial value of only120 mg/l of ammonia. The low concentration of ammonia in AL1 can also be explained in terms of a high rate of biodegradation of waste induced by the high ash fraction and, as a result, ammonia could have been reduced. In AL2, ammonia values remained between those of poor and rich waste and had steeply declined to below the initial concentration by day 121. This pattern remained the same until the last day of monitoring, day 262.

4.3.3 Chloride

Chloride is a conservative parameter which is independent of refuse decomposition and its availability in leachate is mostly dependant on the nature of the waste. In addition, its concentration in water could be increased by a poor water purification system which may result into high levels of chloride since chlorine is used for water treatment. Almost all of the lysimeters show a similar pattern of chloride concentration, especially lysimeters AL1 and AL3. Both lysimeters AL2 and AL3 shows high concentrations of chloride as compared to AL1. A trial test was conducted to determine the initial concentration of chloride in the waste prior the commencement of the experiment. Following the mix design provided in the previous chapter, fresh wastes (i.e. paper, grass and ash components) were used to replicate the waste before the experimental set-up to enable to determine the initial concentration of chloride in the waste. A 50 g sample of dry waste (of all the components mentioned above) representing a particular mix from each lysimeter was prepared. The design of the Orbit Shaker (which is the equipment used for mixing samples for extraction purpose) allows small containers to be used for mixing and as a result, a 50 g sample of dry waste was split into small bottles and a known volume of water was added. It was ensured that all the bottle lids were airtight before the equipment was run. Samples were left for 24 hours mixing and thereafter, the supernatant water in all the bottles were mixed and filtered. Thus, a representative sample was obtained for chloride analysis. The standard MERCK method for chloride analysis used for leachate quality analysis was also applied in this case.

Chloride tests revealed that the type of waste used in the present study contained chloride from the outset. This supports the conclusion that the chloride load in leachate is mostly dependant on the inherent nature of the waste. Test results indicated that lysimeter AL3 (Rich Waste) had the highest initial concentration of chloride, followed by lysimeters AL1 and AL2. The rich waste contained about 7.52 g/kg of chloride at the time of placement, while poor waste contained 3.6 g/kg chloride concentration. The higher initial concentration of chloride in lysimeter AL3 is attributed to high fraction of paper (50 % by dry mass) characterizing rich waste. Since chlorine is used as a bleaching agent in chemical processing of wood pulp for paper manufacturing purpose, chlorides primarily emanate from chlorine used for bleaching. Thus, the increasing paper fraction in waste stream may lead to production of leachates rich in chloride. Lysimeter AL2 initial chloride concentration was found to be 2.72 g/kg. However, chloride in lysimeter AL2 was expected to be higher than that in lysimeter AL1, but less than lysimeter AL3. This is because lysimeter AL2 carried a higher paper fraction than AL1. However, figure 4.5 below shows a sharp decrease in chloride concentration in lysimeters AL1 and AL3 by day 262, while AL2 chloride load increases. This can perhaps be explained in terms of analytical error, but this is unlikely, as the pattern of changing chloride content was similar for all lysimeters, and AL2 consistently showed the highest chloride value. The conclusion that can be safely drawn in this section is that poor waste is characterized by low values of chloride, both from the initial waste before lysimeters had filled and in leachate generated. The reverse is true of rich waste.

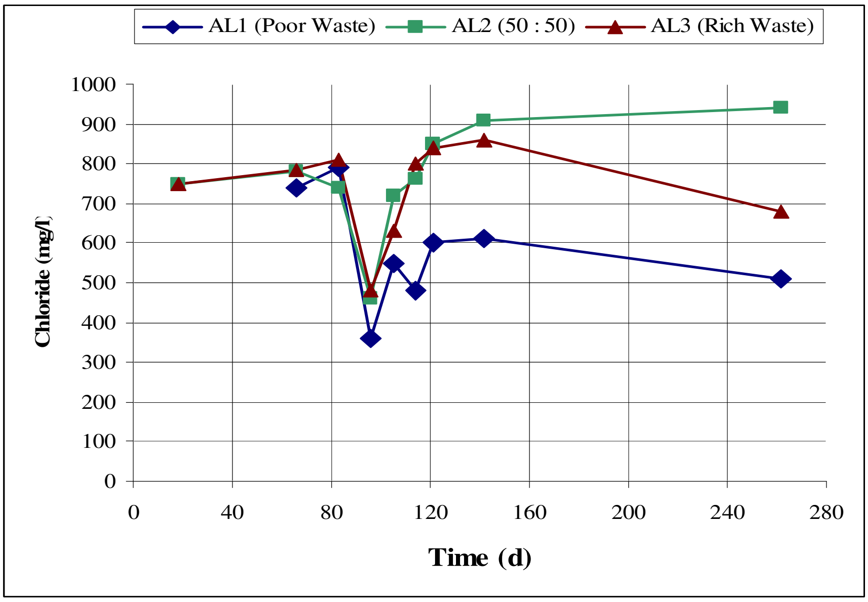


Figure 4.7: Chloride concentrations of leachate for set A lysimeters over the period of monitoring.

4.3.4 Phosphate

Phosphate in the form of phosphorus is one of the major nutrients which facilitate the refuse degradation process. Different cultures of bacteria use it to support the microbial activities, and its concentration also depends on its initial availability (Moola, 2001). In figure 4.8, phosphate values for leachate from lysimeters AL1 and AL2 were very similar from day 83 until day 142. However, by day 262, phosphate concentration in all three lysimeters had returned to their initial concentrations. This can possibly be due to concentrated leachate being collected on day 262 since no water additions had been performed for a prolonged period of time which may have caused dilution of the leachate. However, lysimeters AL2 and AL3 show that phosphate was at least temporarily consumed or used as a substrate by microorganisms to support the waste decomposition process. Lysimeter AL1 remained low in phosphate throughout the test period.

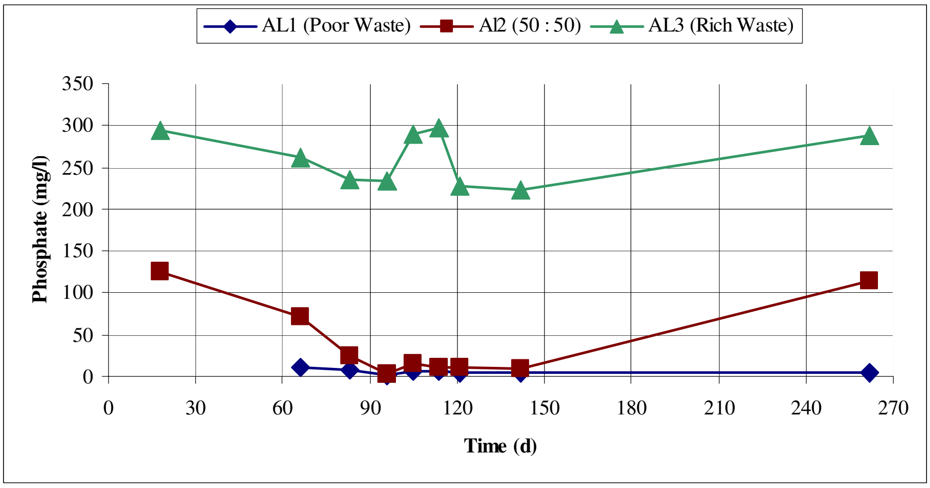


Figure 4.8: Phosphate concentrations for set A lysimeters over the period of monitoring.

4.4 LEACHATE QUANTITY AND QUALITY: SET B LYSIMETERS

4.4.1 Leachate quantity

Water was added to the lysimeters to bring them to their field storage capacities by adding the other half of the field capacity since they were filled at half field capacity and this was done at the end of monitoring programme as stated previously. However, this was not achieved as planned for all the three lysimeters because of water ponding within the lysimeters. Before water could be added, the headspace in each lysimeter was full, which prevented further addition of water. In lysimeters BL1 and BL2, about 2500 ml (2.5 ℓ) of water addition was achieved, while about 2600 ml (2.6 ℓ) was achieved in BL3. Seven days after water was added in the lysimeters, all the lysimeter valves for leachate sampling were opened to collect leachate. No leachate was detected in lysimeter BL3 (rich waste) but only in lysimeters BL1 and BL2. About 253 ml and 358 ml of leachate was released from BL1 and BL2 respectively. Unfortunately, no comparison can be made in terms of leachate generation between BL3 and the other two lysimeters (i.e. BL1 and BL2). The factors which might have led to leachate not being generated in BL3 are not known, but the less pervious nature and high field capacity of the rich waste could perhaps be the major possible factor.

In order to induce synthetic leachate generation in BL3, it was decided to dismantle the lysimeter. After dismantling, quartering was performed to obtain a sample representing the bulk sample of lysimeter BL3 waste. Two samples were prepared, one for leaching and the other one for testing the state of refuse degradation. Leaching was done by placing a known volume of waste into a sieve which rested in an empty basin for leachate collection. This was followed by addition of a known volume of water, and squeezing was done hand as a means of leaching out the contaminants in the rich waste. All the samples obtained from three lysimeters were analysed for leachate quality and the results are presented below.

4.4.2 Leachate quality

Prior to chemical analyses of leachate, samples of leachate were tested for Total Dissolved Solids (TDS). The test was performed in order to determine the amount of dissolved solid matter present in the leachate. It was expected that the hand kneading process might have increased the amount of dissolved solids in leachate, which in turn may affect some of the leachate quality parameters. Total Dissolved Solids (TDS) results were used to ascertain the effects on leachate quality.

The initial step in undertaking this test was to weigh the mass of an empty glass beaker. Secondly, a known volume of leachate sample was added into the weighed beaker and a sample was oven dried for 24 hrs at 55 ^{0}C. After 24 hrs, samples were removed from the oven and the glass beakers were allowed to cool inside a desiccator. Lastly, the beaker with evaporated leachate was weighed in order to obtain the mass of a beaker with residues. Importantly, all the leachate samples were filtered before the test was commenced. The following formula was used to compute the Total Dissolved Solids **(TDS)**:

$$\text{Total Dissolved Solids (g/ℓ)} = 1000 \times \frac{(M_{LE} - M_{TB})}{L}$$

Where,

M_{LE} = mass of beaker after leachate evaporated (in grams)

M_{TB} = tare weight of beaker (in grams)

L = volume of leachate sample (in liters)

Figure 4.9 is a plot for Total Dissolved Solids of leachate for set B lysimeters. Lysimeters BL1 and BL3 are characterized by relatively close amounts of dissolved solids, with BL1 Total Dissolved Solids amounting to 14.8 g/ℓ and 15.2 g/ℓ in BL3. The highest content of Total Dissolved Solids of 55.8 g/ℓ was observed in BL2 as shown in the figure below. The implications of dissolved solids on leachate quality are discussed in the following subsection.

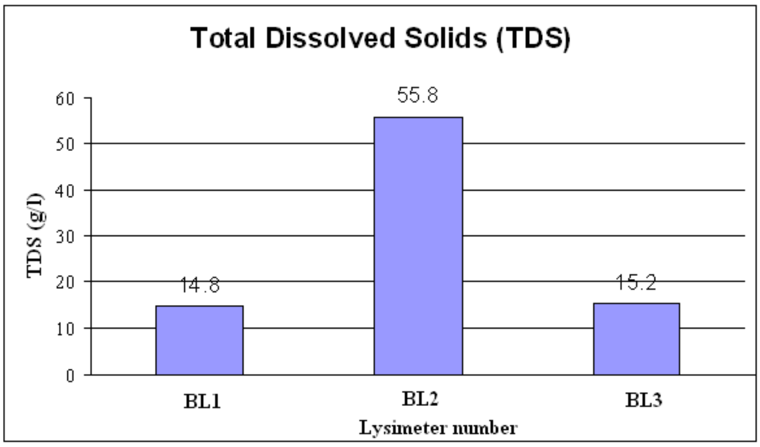


Figure 4.9: Total Dissolved Solids (TDS) in set B lysimeters leachate.

(a) pH and Conductivity

Figure 4.10 shows the relationship between pH and conductivity values for set B lysimeters. Lysimeters BL1 (poor waste) was characterized by a high pH of 8.33 compared to 5.45 and 5.92 in lysimeters BL2 (50: 50) and BL3 (rich waste) respectively. The pH value of 8.33 in BL1 is within the pH range of 7.50 – 9.05 which characterizes leachate quality in AL1 (i.e. set A lysimeters) for the entire period of monitoring. Since both lysimeters AL1 and BL1 were filled with poor waste, it is evident that fly ash which is the highest percentage (60 % dry mass) constituting poor waste, has the neutralizing capacity to produce alkaline conditions. As a result, poor waste has better leachate quality, while acid conditions prevails in 1: 1 (poor/rich) and rich waste lysimeters.

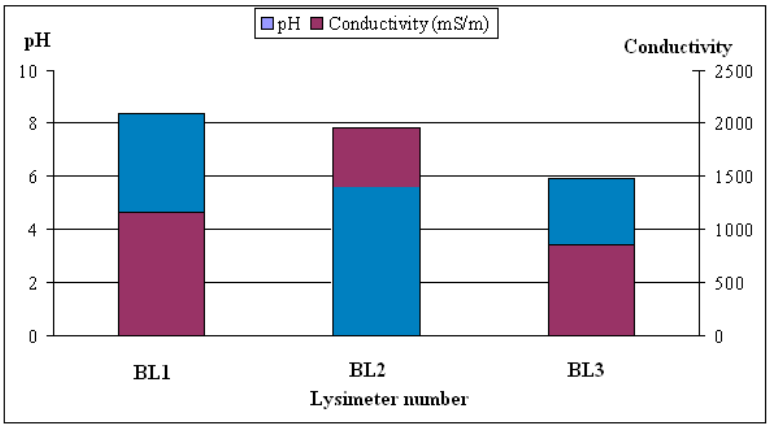


Figure 4.10: Relationship between pH and conductivity values for set B lysimeters leachate.

The relationship between pH and conductivity depicted in figure 10 is inverse, with the lowest pH value of 5.45 in BL2 corresponding to the highest conductivity of 1953 mS/m in the same lysimeter, while BL1 is exhibited by high pH of 8.33 and conductivity of 1158 mS/m. This pattern was also observed in set A lysimeters leachate quality results previously reported in this chapter. However, conductivity value of 1953 mS/m was higher than expected and it was of interest to find out the possible cause/s. As a result, a strong positive correlation was found between conductivity and Total Dissolved Solids (TDS). In figures 4.9 and 4.10, the largest conductivity value of 1953 mS/m coincides with the highest amount of dissolved solids of 55.8 g/ℓ in lysimeter BL2.

(b) Chemical Oxygen Demand (COD)

COD has been used successfully to determine the quality of leachate as stated previously. Set A lysimeter leachate quality results revealed that poor waste exhibited by lower COD values than 50: 50 (poor/rich) and rich waste. This was also observed in COD values of leachate for set B lysimeters as depicted in figure 4.11 below. Lysimeter BL3 (rich waste) attained COD concentration of 66 000 mg/l at the end of the monitoring programme compared to 4990 mg/l and 33 000 mg/l in BL1 and BL2 respectively. These results follow from the lower organic content in the poor waste

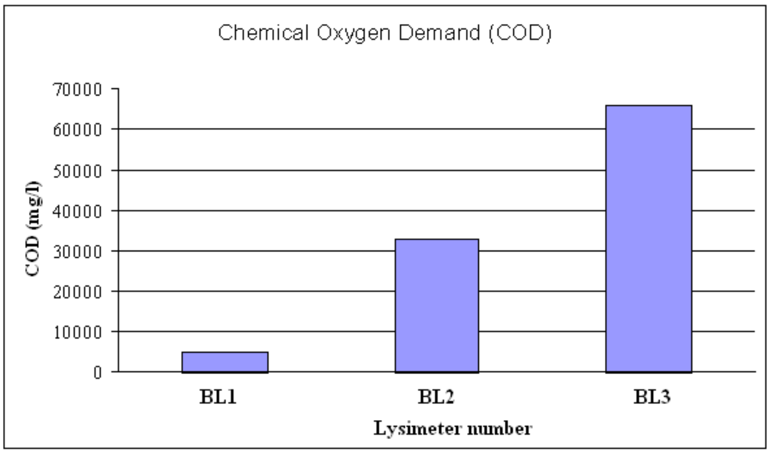


Figure 4.11: COD values for set B lysimeters at the end of monitoring programme.

(c) Ammonia

Figure 4.12 below shows that the ammonia concentrations were high in rich waste and much less in poor waste. It was noted that the ammonia concentration varies as proportions of waste grade from poor to rich. This trend was also noted in set A lysimeters. Lysimeter BL1 (poor waste) exhibited the lowest ammonia concentration of 0.150 mg/l and this perhaps can also attributed to the low organic content of the waste.

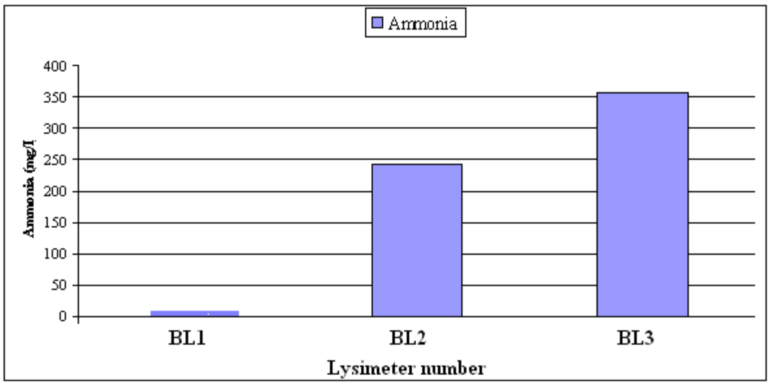


Figure 4.12: Ammonia concentrations for set B lysimeters at the end of monitoring programme.

(d) Phosphate

Figure 4.13 shows measurable concentrations of phosphate for all the lysimeters. Lysimeter BL1 exhibited a phosphate concentration of 1.26 mg/l, increasing to 37.55 mg/l in BL2 and 50.82 mg/l in BL3. Despite the fact that no test for leachate quality was performed in this set of results since no leachate was generated, it is believed that the phosphate concentration is also dependent on the organic content of the waste

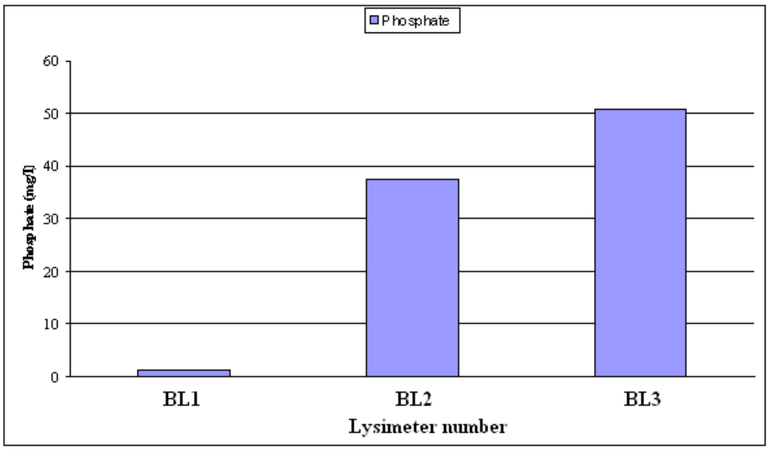


Figure 4.13: Phosphate concentrations for set B lysimeters at the end of monitoring programme.

(e) Chloride

Chloride is a conservative parameter which is independent of refuse decomposition. In addition, its availability in leachate is mostly dependent on the paper content of the refuse. The highest chloride value of 1300 mg/l was observed in lysimeter BL2 (50: 50) as shown in figure 4.14 below. Similarly, lysimeter AL2 (i.e. set A lysimeter) also reflected the same pattern (figure 4.7). Lysimeter BL1 (poor waste) is exhibited by chloride value of 1260 mg/l compared to 850 mg/l in BL3 (rich waste). This trend was unexpected considering the fact that fresh poor waste had an initial low chloride concentration lower than rich waste as observed for set A lysimeter chlorine analysis. Possibly the thorough leaching process used for these lysimeters may have washed out more chloride than leaching test conducted for BL3.

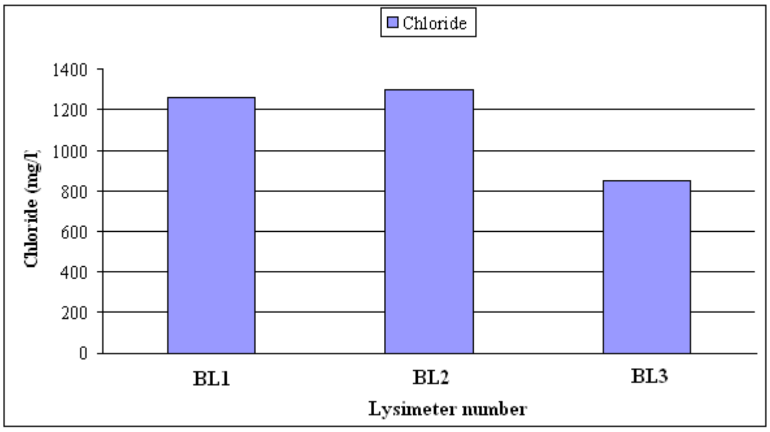


Figure 4.14: Chloride concentrations of leachate for set B lysimeters at the end of monitoring programme.

CHAPTER FIVE: ASSESSMENT OF STATE OF MUNUCIPAL SOLID WASTE DECOMPOSITON

5.1 INTRODUCTION

The decompositional behaviour of waste in a landfill is one of the key aspects to be dealt with during the life of the landfill. It is believed that the extent of refuse degradation should increase as the landfill ages. Recently, some landfills in European countries are being operated as bioreactors in order to enhance the waste decomposition process. One of the advantages claimed for operating a landfill as a bioreactor is that the post monitoring time required after the landfill closed is shortened. However, whether a landfill is operated as a bioreactor or not, research shows that the rate of decomposition can be significantly affected by the waste type disposed of in the landfill.

One of the objectives of this study was to investigate the effect of waste type on waste decomposition. The degree of understanding of the waste decomposition process is of paramount importance, especially prior to commencing of landfill operations in any given climatic zone. For instance, an understanding of the refuse (MSW) degradation process may provide crucial information in regard to refuse degradation rates and landfill gas generation. Such understanding is vital and may affect the costs of operating a landfill significantly if not taken into consideration.

In order to accomplish this specific objective, the two sets of three lysimeters comprising sets A and B lysimeters were dismantled after 262 days of monitoring to assess the extent to which waste composition grading from poor to rich may have affected waste decomposition, specifically for landfills situated on the borderline between B^+ and B^- climatic conditions in South Africa. Therefore, this chapter presents essential steps that had been undertaken during lysimeter dismantling, and on the background of MSW state of degradation followed by the results for examination of the state of waste decomposition.

5.2 DISMANTLING OF THE LABORATORY LYSIMETERS

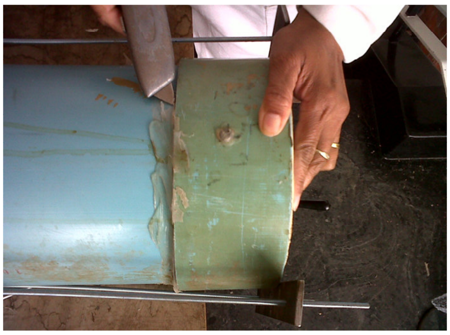
The fundamentals of using the lysimeter to simulate the actual landfill conditions into laboratory is to rapidly provide answers pertaining to landfill processes that could not be readily obtained within a short time framework in a prototype landfill. Despite the fact that most of the conditions in the landfill are of a long term nature, lysimeter studies had shown strong positive correlations to long term results obtained in typical prototype landfill conditions.

Set A lysimeters were dismantled after 262 days of monitoring in order to assess the extent to which varying proportions of paper, ash and putrescible content may affect waste decomposition. Lysimeter dismantling had been carried out to obtain representative samples to be used for determining the state of waste decomposition, as well as for determining the final moisture contents of waste in individual lysimeters.

Leachate was collected from set A lysimeters (i.e. lysimeters AL1, AL2 and AL3) before the lysimeters were dismantled on the same day. Samples were tested immediately for leachate quality. After leachate had been collected, the lysimeters were unwrapped from the bubble-wrap insulation. This was followed by unscrewing the rods that secured the plates at the two ends of the lysimeter. A knife was used to remove the silicon sealer from all the joints, and a hammer used to release the plates closing the bottom and top of the lysimeter. Extruding waste from the lysimeters was a slow and tedious operation. It was important to take measures before-hand to avoid sample spillage and contamination. After the two plates closing the top and bottom of the lysimeter were removed, the lysimeter was laid horizontally on a table for sample extrusion. First, the sand drainage layer at the bottom of the lysimeter was removed and the bottom part of the lysimeter was covered with a plastic bag for sample collection. Due to the relatively high density of confined compacted waste within the lysimeter, a compaction hammer had to be used to mechanical extrude the waste. The compaction hammer was placed within the top part of the lysimeter against to the wall and the lysimeter was pushed against the wall from the opposite direction where the plastic bag for sample collection is placed (see figure 5e). Owing to the reactive force exerted by the wall on the compaction hammer, the compaction hammer pushed waste into the plastic bag. Sampling was accomplished in this way and the plastic bag was sealed immediately to avoid contamination and moisture loss. This process was repeated for each lysimeter. A summary of the major steps followed for lysimeter dismantling is shown in figures 5a to f below.

a) Undoing the rods tightening the lysimeter plates

b) Removal of the bubble-wrap



c) The use of knife to remove silicon sealer at the conjunctions

d) Removal of sand layer for drainage at the base of the lysimeter

e) Pushing the lysimeter against the wall for sample extrusion

f). Bulk sample of waste obtained after dismantling lysimeter

Figure 5.1 (a, b, c, d, e & f) shows major steps followed during lysimeter dismantling.

5.3 MEASURING THE STATE OF DEGRADATION OF THE MSW

A range of methods has been used to assess the state of MSW degradation. The application of these methods depends on the parameters to be analysed, for example, total carbon (TC) is used to determine the carbon content of organic matter present in the waste; loss on ignition (LOI) was used to measure the combustibility of waste and the Biochemical Methane Potential (BMP) test for estimation of methane production potential. The latter has received attention for estimating the degradability of the organic matter by measuring the volume of methane produced by a sample of waste incubated under controlled anaerobic conditions (Richards et al., 2005). The major drawback of this method is its slowness and the care required to ensure that anaerobic conditions are maintained throughout the test. However, it provides reliable results. Several methods from other fields of study such as agricultural sciences have also been adopted in order to test the state of waste decomposition. These include: Neutral Detergent Fibre (NDF), Acid Detergent Fibre (ADF) and Acid Digestible Lignin (ADL) tests. These tests collectively have been routinely used to determine the fibre concentration of forage and animal feeds and the details for these tests are provided by van Soest et al., 1991. Briefly, the NDF test has the capability to dissolve easily degradable material such as pectins, sugar starch and fats. Most importantly, this method leaves behind essential cell wall components of plant material such as cellulose, hemicellulose and lignin. On the other hand, the ADF test digests less degradable hemicellulose and some proteins leaving behind cellulose, lignin and bound nitrogen as residues, whereas the ADL test is a method commonly used to determine the lignin content. These tests are directly used to measure the organic fraction of MSW which is mainly composed of cellulose (C), hemicellulose (H) and lignin (L). The relative content of C, H and L has been successful used to predict landfill degradation rates and methane quantities due to their susceptibility to decomposition. For this reason, similar tests were conducted in the present study by the Agriculture Research Council (ARC-LNS) to aid understanding of the decomposition of waste. Richards et al. (2005) has provided a summary of the essential components that are determined by these tests and they appear below as follows:

NDF = Cellulose + Hemicellulose + Lignin + Ash

ADF = Cellulose + Lignin + Mineral Ash

ADL = Lignin + Mineral Ash

Hence:

Cellulose content = ADF – ADL = C

Hemicellulose content = NDF – ADF = H

Lignin content = ADL = L

5.3.1 Summary of municipal solid waste characterization into cellulose, hemicellulose and lignin

Several studies have been undertaken to characterize MSW composition in terms of cellulose, hemicellulose and lignin. Tables 5.1 and 5.2 below summarize a range of values of cellulose, hemicellulose and lignin content found in MSW. It is worthwhile to mention that table 5.1 shows overall values for different streams of fresh MSW disposed of in a particular landfill site, while table 5.2 depicts contents of cellulose, hemicellose and lignin for different waste types. Both tables show a large variety of values and this is attributed to differences in composition and quantities of municipal solid waste between regions, states and countries (Lamborn, 2009). These values can be used as a guide for modeling landfill waste degradation rates and gas generation. The average cellulose values in both tables exceed those of hemicellulose and lignin, and the cellulose content in MSW accounts for 90 % of methane generation in the landfill (Barlaz, 2006).

Table 5.2: Characterization of MSW into cellulose, hemicellulose and lignin and their ratios: a comparative study (Lamborn, 2009).

Cellulose C (%)	Hemicellulose H (%)	Lignin L (%)	$\frac{(C+H)}{L}$	Source
17.28	5.14	12.67	1.77	Ham, Norman et al. 1993; Eleazer, Odle lii et al. 1997
28.8	9	23.1	1.64	Barlaz, Eleazer et al. 1997b; Barlaz, Eleazer et al. 1997a
42.4	6.6	10.9	3.89	Barlaz 2006; Ham

				and Bookter 1982
25.6	11.9	7.2	4.5	Jones and Grainger (1983a, b)
63.4	9	15.7	4.04	Bookter and Ham 1982
51.2	8.7	15.2	4.15	Barlaz et al., 1989
28.8	10.6	23.1	1.64	Eleazer et al., 1997
38.5	6.7	28	1.68	Rhew and Barlaz 1995
48.2	10	14.5	4.06	Ress et al., 1998
36.7	10.8	13.6	3.19	Barlaz (unpublished)
43.9	5.8	25.1	2.15	Price et al., 2003
54.3		12.1	5.38	Barlaz (unpublished)
22.4		11	2.57	Barlaz (unpublished)
24.8	6.7	9.7	3.2	Ivanova, Richards et al., 2007a

Table 5.2: Characterization of different waste type in cellulose, hemicellulose and lignin and their ratios: a comparative study (Lamborn, 2009).

Waste type	Cellulose C (%)	Hemicellulose H (%)	Lignin L (%)	$\frac{(C+H)}{L}$	Source
Newspaper	51		2.5		Stinson and Ham 1995
	48.5	9	2.9	2.41	Eleazer, Odle lii et al. 1997
	18.5	9			Environmental _ Agency 2004
	48.3	18.1	22.1	3.00	Wu et al. 2001; Barlaz 2006
	48.5	9	23.9	2.41	Eleazer et al., 1997
	48.5	9	23.9	2.41	Barlaz 2008
Office paper	87.4	8.4	2.3	41.65	Eleazer, Odle lii et al. 1997
	64.7	13	0.93	83.55	Wu et al. 2001; Barlaz 2006
	87.4	8.4	2.3		Eleazer et al., 1997
	87.4	8.4	2.3		Barlaz 2008
Grass	26.5	10.2	28.4	1.29	Eleazer, Odle lii

					et al. 1997; Barlaz, Eleazer et al. 1997b
	25.6	14.8	21.6	1.87	Barlaz, Eleazer et al. 1997a
	26.5	10.2	32.6	1.13	Barlaz 2008

5.4 ANALYSIS AND DISCUSSION OF RESULTS

The bulk samples of waste obtained from the lysimeters after dismantling were thoroughly mixed and a sample was taken and oven dried at 55 ^{0}C in order to determine moisture contents, as well as for testing the state of refuse degradation. Samples were oven dried at 55 ^{0}C for at least 24hrs in order to obtain relatively constant mass before they were sent to the laboratory for analysis of parameters used to determine the state of waste decomposition. Basically, three parameters, namely cellulose, hemicellulose and lignin were chosen to provide an insight into degradation of MSW. The specifications and methods (i.e. NDF, ADF and ADL) used to determine these parameters have not been disclosed by the laboratory.

The analysis of results for these parameters is difficult to interpret unless their initial state in the waste is known. For this reason, the waste mix used in this study replicated the synthetic MSW initially used to fill the individual lysimeters. All the samples representing the waste before the lysimeters were filled and waste after lysimeter dismantling were sent to the laboratory for analysis of the same parameters. This following section compares the results of cellulose, hemicellulose and lignin contents of fresh and decomposed waste.

Set A lysimeters

Figure 5.2 shows cellulose values for waste grading from poor to rich; before and after decomposition. Although there was no test in the present study conducted to initially characterize different waste compositions (Poor Waste; 50: 50 and Rich Waste) by cellulose, hemicellulose and lignin content, it is believed that the cellulose content results reported in this section originate from the paper (i.e. newspaper and office paper) and grass fractions used in this study, and Table 5.2 has been used as guideline. The cellulose content for fresh waste in lysimeter AL1 (poor waste) is less than in lysimeters AL2 (50:

50) and AL3 (rich waste) cellulose content. This can be explained in terms of the increase in paper and grass fractions in lysimeters AL2 to AL3 compared to AL1. The cellulose content in MSW is affected by organic fraction quantity as shown in the figure below. Figure 5.2 shows a decrease in cellulose content in all three lysimeters after decomposition had taken place, with the largest decrease of 19.67 % in lysimeter AL1. The significant change in cellulose content in lysimeter AL1 shows a higher degree of waste degradation as compared to lysimeters AL2 and AL3 where the decrease was 9.7 % and 8.85 % respectively. The high degree of degradation in lysimeter AL1 is attributed to the high ash fraction which characterizes the poor waste. This indicates that fly ash has the capacity to enhance refuse degradation and as a result, the methanogenic phase is also thought to have been established earlier in AL1 than in lysimeters AL2 and AL3.

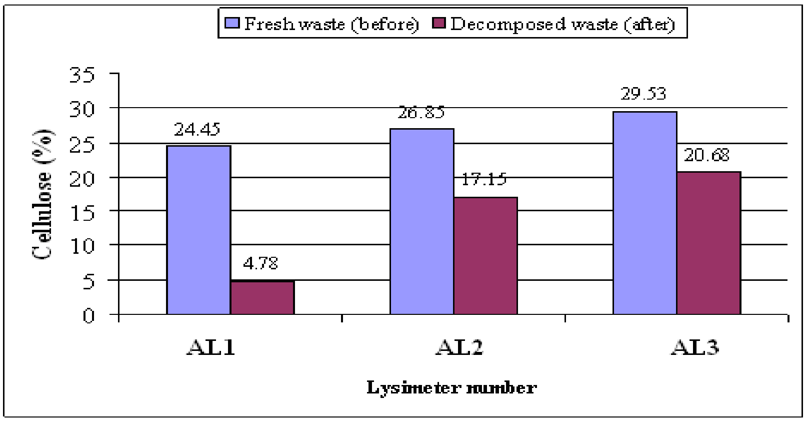


Figure 5.2: Cellulose content of MSW in set A lysimeter before and after decomposition.

There is a slight difference in cellulose content in lysimeters AL2 and AL3 after decomposition had taken place, with AL2 cellulose content decreasing from 26.85 – 17.15 (= 9.70 %), while in AL3 cellulose content decreased from 29.53 – 20.68 % (8.85 %). However, there is a considerable gap in cellulose values of degraded waste between AL1 and AL2, especially in AL3. This shows that cellulose fraction in poor waste degrades more than rich waste, especially as initial cellulose values were approximately the same. This significant difference in degradation rates can also be expected in prototype landfills receiving poor and rich waste. As a result, landfills receiving poor and rich waste may need special attention in terms of design and/or operation by also considering other factors such leachate quality produced by these two types of waste as discussed in the previous chapter.

Figure 5.3 shows hemicellulose values of MSW in A lysimeters and the values reported here also fall within the previously reported data range (see Tables 5.1 and 5.2). The hemicellulose content is less degradable as compared to cellulose. A slight change in hemicellulose content in fresh waste is noted, with hemicellulose values decreasing from poor to rich waste. Lysimeters AL2 and AL3 show an increase in hemicellulose content after degradation had taken place. However, it was expected that hemicellulose would decrease as degradation took place but samples from AL2 and AL3 recorded a slight increase. There are three possible reasons for this difference: 1). analytical error, 2). slight differences in hemicellulose content in the initial and degraded samples, 3). Lysimeter AL1 started at slightly higher hemicellulose content than lysimeters AL2 and AL3, and a decrease in hemicellulose content was noted only in AL1. This is an indication that more degradation had taken place in AL1 as compared to lysimeters AL2 and AL3.

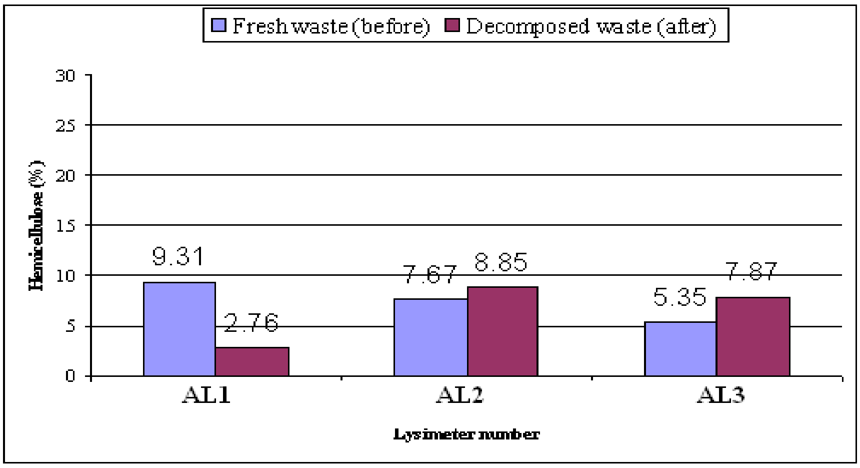


Figure 5.3: Hemicellulose content of MSW in set A lysimeter before and after decomposition.

In figure 5.4 below, lignin content is attributed to both the paper and grass fractions used in the present study. This figure shows large discrepancies in lignin content of MSW before and after decomposition in lysimeters AL2 and AL3. This is difficult to explain. Research (Moola, 2001) shows that this parameter does not readily degrade under anaerobic conditions. However, lysimeter AL1 (poor waste) indicates that the lignin had degraded slightly as compared to cellulose and hemicellulose.

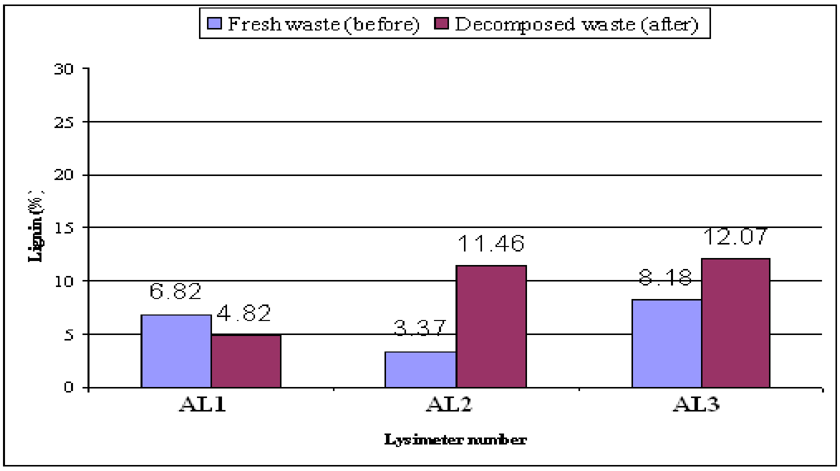


Figure 5.4: Lignin content of MSW in set A lysimeters before and after decomposition.

To supplement the refuse degradation data present above, the $\frac{(C+H)}{L}$ ratios for waste before and after decomposition were computed in order to assess the extent of refuse degradation (see figure 5.5 below). The $\frac{(C+H)}{L}$ ratio in poor waste (AL1) decreased from 4.95 to 1.56 %, while AL2 (50: 50) ratio started at 10.24 % and ended up at 2.67 % after lysimeter dismantling. Lastly, AL3 ratio varied from 4.26 to 2.37 %. In conclusion, poor waste showed better performance in decompositional behaviour as compared to rich waste. However, the findings in this section are constrained to one method and only give an insight on decompositional behaviour of waste grading in composition from poor to rich waste.

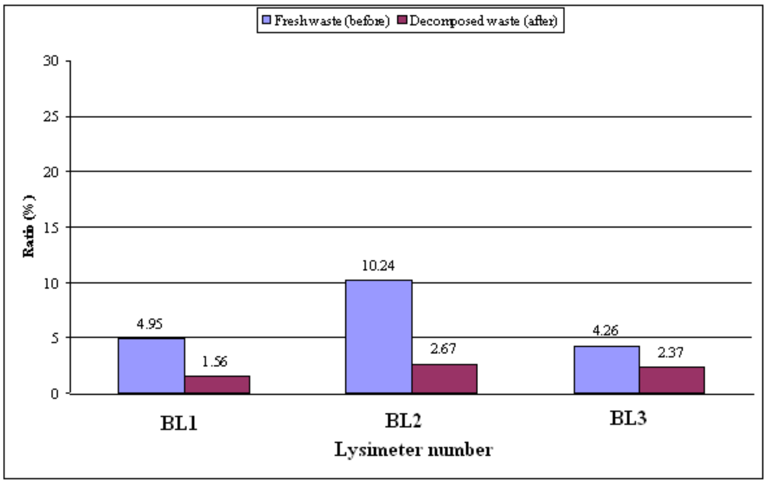


Figure 5.5: $\frac{(C+H)}{L}$ **ratios of fresh and decomposed waste in set A lysimeter.**

Set B lysimeters

It is significant to mention that the difference between set A and B lysimeters was that set A lysimeters were filled with waste at field storage capacities and leachate was circulated during the test period, while set B lysimeters were filled at half field storage capacities without being irrigated during the test period. Both sets were filled with waste of similar characteristics grading from poor to rich waste. It was of interest at the end of the experiment to dismantle both sets of lysimeters to enable the comparison of their state of degradation resulting from difference in water contents in these two sets. This section presents result for cellulose, hemicellulose and lignin content in set B lysimeters.

Figure 5.6 shows cellulose content of fresh and decomposed MSW in set B lysimeters. Both sets (A and B) are characterized by similar initial content of cellulose, hemicellulose and lignin in fresh waste. The low cellulose content of fresh waste in lysimeter BL1 (poor waste) as compared to lysimeters BL2 (50 : 50) and BL3 (rich waste) is attributed to the increase in paper and grass fractions in lysimeters BL2 and BL3 compared to BL1 as discussed in the previous section. Figure 5.6 shows a decrease in cellulose content in all three lysimeters after decomposition had taken place, with the higher decrease of 9.98 % (24.45 - 14.47 %) in BL1 compared to 6 % (26.85 - 20.85 %) and 2.61 % (29.53 - 26.92 %) in lysimeters BL2 and BL3 respectively.

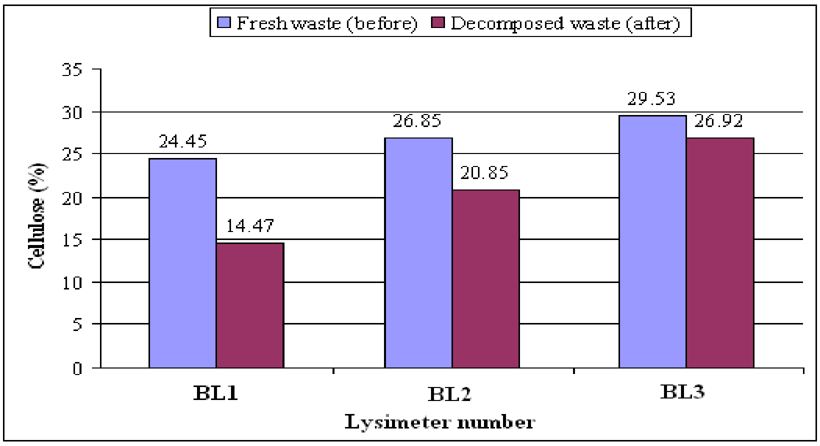


Figure 5.6: Cellulose content of MSW in set B lysimeter before and after decomposition.

The difference in cellulose content especially in BL1 and BL3 shows that cellulose fraction in poor waste degrades more than rich waste and this is attributed to the high ash fraction characterizing the poor waste. A similar pattern was also noticed in set A lysimeters. It is important to mention that the cellulose content in set A lysimeters degraded more than in set B lysimeters (see figures 5.2 and 5.6). Lysimeter AL1 (poor waste) in A lysimeters shows a difference in cellulose degradation of 9.69 % (14.47 – 4.78 %) compared to BL1 (poor waste) in B lysimeters after the decomposition had taken place. This trend is also common in AL2 (50: 50) and AL3 (rich waste), with a decrease in cellulose content of 7.70 % (20.85 – 17.15 %) in AL2 more than in BL2. The least cellulose content decrease of 2.61 % of decomposed MSW was noted in lysimeter BL3. Since both set A and B lysimeters were filled with waste of similar compositions at varying proportions, it is apparent that the difference in degree of waste decomposition between these two sets based on the cellulose content results reported in this section, has been induced by the various moisture contents which characterize the waste specimens.

The pattern of hemicellulose content of fresh and decomposed waste is depicted in figure 5.7 below. Lysimeter BL1 (poor waste) shows a decrease of hemicellulose content of 4.25 % (9.31 – 5.06) and this marks the degree of degradation in this particular lysimeter. On the other hand, AL1 (poor waste) in set A lysimeters exhibited a 6.55 % (9.31 – 2.76 %) decrease of hemicellulose content as compare with BL1. It was difficult to explain the pattern of hemicellulose content changes in set A lysimeters (AL2 and

AL3) because of the higher hemicellulose content in decomposed waste than fresh waste. However, BL2 showed a slight decrease in hemicellulose content of 0.42 % (7.67 – 7.25 %) and this indicates that 1: 1 (poor/rich) had degraded hemicellulose much less than poor waste. Lysimeter BL3 (rich waste) is characterized by a higher cellulose content in decomposed waste than in fresh waste. Three possible reasons for the hemicellulose change pattern had been suggested for set A lysimeters hemicellulose content, specifically lysimeters AL2 and AL3.

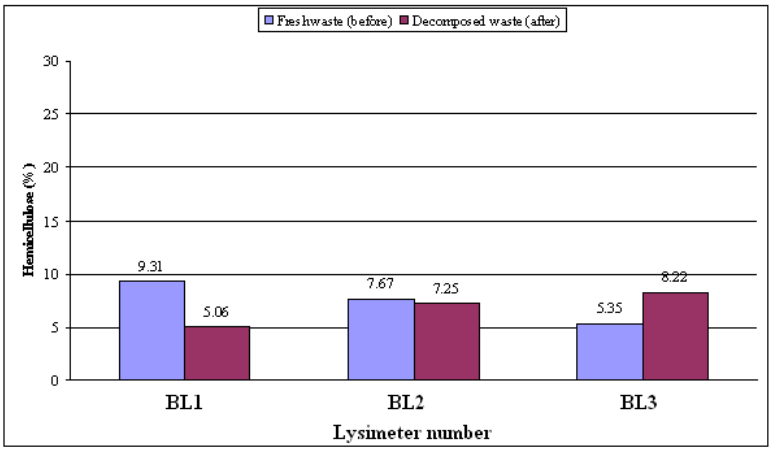


Figure 5.7: Hemicellulose content of MSW in set B lysimeter before and after decomposition.

Figure 5.8 below shows the complexity of lignin degradation pattern due to high lignin content of decomposed MSW exceeding that of fresh waste in lysimeters BL2 (50: 50) and BL3 (rich waste). This pattern is difficult to explain as it was in set A lysimeters. However, BL1 (poor waste) shows a slight decrease of 0.55 % (6.82 – 6.27 %). The slight decrease in lignin content was also noted in AL1 (poor waste) set A lysimeter.

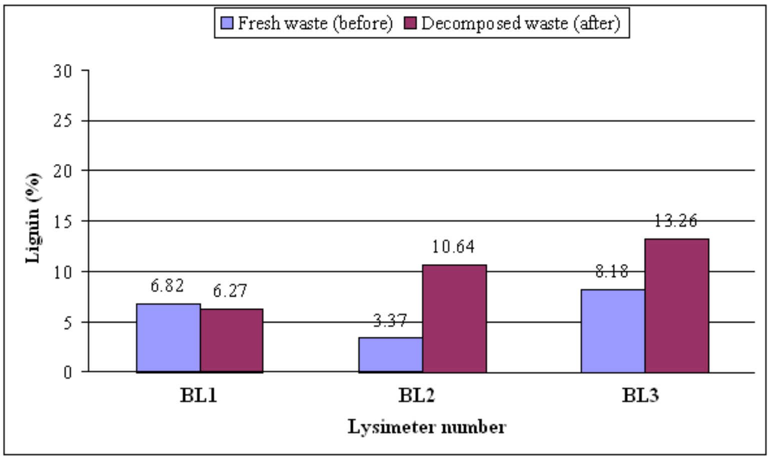


Figure 5.8: Lignin content of MSW in set B lysimeters before and after decomposition.

A comparison of $\frac{(C+H)}{L}$ ratios was made to aid understanding the degree of refuse degradation before and after as it appears in figure 5.9 below. The $\frac{(C+H)}{L}$ ratio for BL1 (poor waste) was 4.95 % before decomposition and about 3.11 % after decomposition had taken place. Lysimeters BL2 (50: 50) and BL3 (rich waste) shows low ratios compared to BL1 (poor waste) which had performed better in terms degradation of cellulose, hemicellulose and lignin contents than 1: 1 (poor/rich) and rich waste. The high lignin content of decomposed MSW was found to be the main factor accountable for low $\frac{(C+H)}{L}$ ratios in BL2 and BL3. Hence, the source and/or factors affecting the high lignin content of decomposed waste in both set A and B cannot be explained.

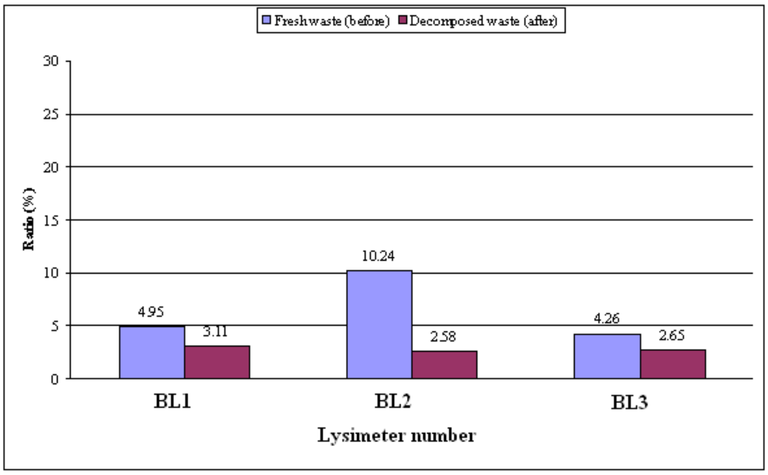


Figure 5.9: $\frac{(C+H)}{L}$ **ratios of fresh and decomposed waste in set A lysimeter.**

In conclusion, poor waste in both lysimeter sets is characterized by a higher degree of decomposition for organic waste than poor/rich and rich waste and this is attributed to high ash fraction in poor waste. The degree of degradation of poor waste in both sets was found to be different, with a high rate of degradation occurring in AL1 set A lysimeter (where irrigation of lysimeters had taken place during the time of monitoring). BL1 exhibited a low rate of waste decomposition. It can be safely concluded that initial water content greatly affects waste decomposition rate.

CHAPTER SIX: SUMMARY, CONCLUSIONS AND RECOMMENDATIONS FOR FURTHER STUDIES

6.1 SUMMARY

The present study set out to investigate the leachate generation potential from landfills situated in two different climatic conditions, namely B^+ (water surplus) and B^- (water deficient), as well as the decomposition of refuse (MSW) having different proportions. Three components of wastes: paper, grass and fly ash with varying proportions, were used to model waste from “poor” and “rich” suburbs. A total of six lysimeters were used in the present study. Lysimeters were split into two sets (i.e. A and B) and were filled with waste grading from poor to rich at half and full field capacities. With set A lysimeters modeling B^+ (water surplus) landfills, while set B^- (water deficient) landfills. The leachate from lysimeters at field storage capacities was recycled and leachate generation data was recorded. No leachate was generated in set B lysimeters during the entire period of monitoring. However, the waste from set B lysimeters was leached at the end of the test to produce leachate. The leachate generated from the lysimeters was analysed for several parameters to determine the quality of leachate. This has enabled the effect of waste type on leachate quality to be examined. Lastly, the extent of decomposition of MSW was investigated in both lysimeter sets.

The cumulative leachate generation in lysimeter AL1 (poor waste) was low as compared to leachate generation in lysimeters AL2 and AL3 but the field capacity was also low. Only the lysimeters filled at full field capacity generated leachate. The details of leachate generation data revealed that leachate generation is affected by waste type as well as water volumes. It has also been noted that if these factors are considered and others such as climate, two landfills one receiving poor waste and other rich waste, may generate distinctively different volumes of leachate. This was noted in lysimeter AL1 filled with poor waste generating 60 % of leachate generated by a lysimeter filled with 50 % poor and 50 rich waste and with a lysimeter filled with rich waste, although three had been filled at field capacity. However, the field of poor waste was 70 % of that for 1: 1 waste.

An investigation of the effect of waste type on leachate quality was also undertaken. This included analysis of several parameters used to determine the quality of leachate, namely pH, conductivity, COD, chloride, ammonia and phosphate. The lysimeters filled with poor waste (AL1 and BL1) were characterized by low values of these parameters as compared to concentrations in leachate from lysimeters AL2/BL2 (poor/rich waste) and AL3/BL3 (rich waste) in both sets. The "better" leachate quality characterizing lysimeter AL1/BL1 (poor waste) was attributed to the high fly ash content (60 % by dry mass) which shows high neutralizing capacity as compared to AL2/BL2 (poor/rich) and AL3/BL3 (rich waste) lysimeters. Lysimeters AL3/BL3 (rich waste) showed "poor" leachate quality, and low pH values revealed that strong acidic conditions prevailed in these lysimeters throughout the tests. As a result, some of the conditions required for establishment of methanogens might not have existed or were inhibited by the acid conditions. Hence, the concentrations of the above mentioned parameters in lysimeters AL2/BL2 (50: 50) lay between those of AL1/BL1 and AL3/BL during the time of study. These findings show that leachate collection and management systems of landfill disposing of poor waste may be operated at a lower cost than a landfill receiving rich waste. The costs may be significantly reduced by means of no need or less demand to invest in the leachate treatment facility because of the low leachate strength characterizing poor waste.

Lastly, the effect of waste type on decompositional behaviour of MSW was investigated. The prime aim of this investigation was to assess the extent to which waste with varying proportions of paper, grass and fly ash may affect waste decomposition. Three parameters, namely the contents of cellulose, hemicellulose and lignin were analysed to assess the degree of decomposition of fresh and decomposed MSW. The poor waste showed a better performance in terms of degradation of cellulose, hemicellulose and lignin than lysimeters AL2/BL2 (poor/rich waste) and AL3/BL3 (rich waste). It is evident from the results that poor waste can degrade more easily than rich waste, as noted from their degradation rates. Since no irrigation took place in set B lysimeters, it was of interest to assess the effects of water content on waste decomposition. Results revealed that poor waste performed better in terms of refuse degradation in a water deficient environment than 1: 1 (poor/rich) and rich waste.

6.2 CONCLUDING REMARKS

Several tests have been conducted in the present study in order to obtain sufficient data to test the validity of the hypothesis made. The analysis and discussion of results presented in the previous chapters has led to the following conclusions:

- There is a potential for leachate generation from landfills situated in B^+ sites. Leachate generation is a factor of waste type and water volumes added. Poor waste generates more leachate than 50 % poor and 50 % rich waste, as well as rich waste due to its low field storage facility, under similar climatic conditions.

- Poor waste is characterized by good leachate quality based on low concentrations of COD, Conductivity, pH, Ammonia, Chloride and Phosphate as compared to rich waste. This is due to the neutralizing capacity of the 60 % fly ash content.

- Fly ash plays a crucial role in the process of waste decomposition. Thus, poor waste is characterized by a higher rate of degradation than the fly ash deficient rich waste.

- The water content affects the rate of refuse degradation in landfills.

- It is believed that a landfill receiving either poor or rich waste only under the same climatic conditions may require different standards for waste disposal by landfilling. A relaxation of Minimum Requirements for landfills disposing poor waste could significantly reduce the cost of establishing a landfill under a range of climatic conditions than landfills receiving rich waste.

The conclusions are applicable in cases of the landfills receiving three components of waste used in the present study under similar climatic conditions. However, these conclusions must be exercised with caution and only be used as a guideline.

6.3 RECOMMENDATIONS FOR FURTHER STUDIES

The present study investigated the potential for leachate generation from landfills situated in B^+ (water surplus) and B^- (water deficient) and decomposition of MSW under these conditions by means of simulated laboratory lysimeters. The study of this nature was the first to be conducted for landfills situated in B^+ and B^- climatic zone. As a result, this work can be expanded from not using laboratory lysimeters only, but to the implementation of intensive field survey tests. Based on the observations made from the study, the following recommendations for future studies were made:

- Identification of landfill sites situated in two climatic zones, namely B^+ and B^- to comprehensively study the water balance measurements and leachate quality in order to evaluate the costs to operate the landfill in particular climatic conditions (for at least 5 years). This may also help to evaluate the need for special regulatory framework for landfills situated in the borderline between B^+ and B^-.

- Gas analysis measurements must be undertaken both in laboratory and in the field in order to assess the effect of waste type on gas generation.

- Assessment of MSW degradation must be supplemented by other methods such as Biochemical Methane Potential Assays to improve understanding of refuse decomposition process for landfills situated between B^+ and B^- climatic conditions.

APPENDICES

APPENDIX A: PHYSICAL PROPERTIES OF WASTE

Moisture content determination:

$$W = \frac{Ww}{Ws} \text{ x } 100$$

Where:

W = Water content (%)

Ww = Weight of water present in the waste

Ws = Weight of dry waste

Initial and final moisture contents of MSW, as well as their field storage capacities

Lysimeter numbers	Initial moisture content in dry mass basis (%)	Final moisture content in dry mass basis (%)	Field storage capacity in dry mass basis (%)
AL1	1.43	156	168
AL2	1.47	154	241
AL3	1.50	244.61	316

APPENDIX B: WATER BALANCE MEASUREMENTS

C1: Total water added and leachate generated from set A lysimeters

Lysimeter number	Total water added (ml)	Total leachate generated (ml)
AL1	49.94	456.82
AL2	690.56	539.86
AL3	422.43	277.94

C2: Water additions and leachate generated from individual set A lysimeters

Lysimeter AL1

Day	66	83	96	105	114	121	142
Water added (ml)	84	43.58	88.54	94.88	94.35	64.37	71.1

Day	66	83	96	105	114	121	142
Leachate generation (ml)		43.58	88.54	94.88	94.35	64.37	71.1

Lysimeter AL2

Day	66	83	96	105	114	121	142
Water added (ml)	227.8	174.8	34.4	103.15	87.87	61.54	77.1

Day	66	83	96	105	114	121	142
Leachate generated (ml)		174.8	34.4	103.15	87.87	61.54	77.1

Lysimeter AL3

Day	66	83	96	105	114	121	142
Water added (ml)	177.24	45.4	31.33	76.08	63.08	29.3	32.75

Day	66	83	96	105	114	121	142
Leachate generated (ml)		45.4	31.33	76.08	63.08	29.3	32.75

APPENDIX C: LEACHATE QUALITY DATA

Set A lysimeters

Lysimeter AL1 leachate quality data

Day	pH	Conductivity (mS/m)	COD (mg/l)	NH_4 (mg/l)	Cl (mg/l)	PO_4 (mg/l)
0	0	0	0	0	0	0
18						
66	7.50	1451	20500	110	740	11.18
83	7.85	884	15700	72.674	790	7.35
96	7.72	785	23600	8.65	360	2.25
105	7.78	781	15000	10.024	550	5.51
114	7.88	731	3020		480	6
142	7.92	705	2820	0.6	600	5
262	9.05	528	1510	0.80192	510	4.4902

Lysimeter AL2 leachate quality data

Day	pH	Conductivity (mS/m)	COD (mg/l)	NH_4 (mg/l)	Cl (mg/l)	PO_4 (mg/l)
0	0	0	0	0	0	0
18	6.03	1425	22432	168	750	125.52
66	5.7	1625	17900	499	784	72
83	6.38	1611	38800	383.42	810	25
96	7.55	1529	25800	338.31	480	3.61
105	7.28	1496	53300	415.37	630	15
114	7.31	1119	9310	163	800	11
142	8.02	955	5690	126.55	840	11.4
262	8.4	725	11000	156.625	920	114.099

Lysimeter AL3 leachate quality data

Day	pH	Conductivity (mS/m)	COD (mg/l)	NH_4 (mg/l)	Cl (mg/l)	PO_4 (mg/l)
0	0	0	0	0	0	0
18	5.2	1512	24077.25	294	750	294
66	5.1	1617	19700	576.38	780	261
83	5.07	1670	33500	620	740	235.32
96	5.09	1642	23900	608	460	234.52
105	5.08	1646	71500	796	720	290.23
114	5.10	1600	22800	661	760	297.20

142	5.11	1607	11350	603.51	850	227
262	5.11	1618	11410	545.6815	680	288.1892

Set B lysimeters

Lysimeter BL1 leachate quality data

Day	pH	Conductivity (mg/l)	COD (mg/l)	NH_4 (mg/l)	Cl (mg/l)	PO_4 (mg/l)
262	8.33	1158	4990	0.150	1260	1.255215

Lysimeter BL2 leachate quality data

Day	pH	Conductivity (mg/l)	COD (mg/l)	NH_4 (mg/l)	Cl (mg/l)	PO_4 (mg/l)
262	5.45	1953	30800	243.33	1300	37.5544

Lysimeter BL3 leachate quality data

Day	pH	Conductivity (mg/l)	COD (mg/l)	NH_4 (mg/l)	Cl (mg/l)	PO_4 (mg/l)
262	5.92	850	66000	357.23	850	50.821

APPENDIX D: STATE OF MUNICIPAL SOLID WASTE DEGRADATION

Methods used:

NDF: Neutral Detergent Fibre

ADF: Acid Detergent Fibre

ADL: Acid Digestible Lignin

The components that are determined by these tests are:

NDF = Cellulose + Hemicellulose + Lignin + Ash

ADF = Cellulose + Lignin + Mineral Ash

ADL = Lignin + Mineral Ash

Computation of cellulose, hemicellulose and lignin contents:

Cellulose content = ADF – ADL

Hemicellulose content = NDF – ADF

Lignin content = ADL

Municipal Solid Waste Degradation Data

Set A lysimeters

Waste Degradation Parameters (%)	**Fresh Waste (Initial)**			**Decomposed Waste (After)**		
	A/L1	**A/L2**	**A/L3**	**AL1**	**AL2**	**AL3**
NDF	40.45	37.89	43.05	12.36	37.46	40.62
ADF	31.14	30.22	37.71	9.60	28.61	32.75
ADL (Lignin)	6.82	3.37	8.18	4.82	11.46	12.07
Cellulose	24.45	26.85	29.53	4.78	17.15	20.68
Hemicellulose	9.31	7.67	5.34	2.76	8.85	7.87
$\frac{(C+H)}{L}$ **ratio**	4.95	10.24	4.26	1.56	2.67	2.37

Set B lysimeters

Waste Degradation Parameters (%)	Fresh Waste (Initial)			Decomposed Waste (After)		
	B/L1	**B/L2**	**B/L3**	**BL1**	**BL2**	**BL3**
NDF	40.45	37.89	43.05	25.80	37.55	48.40
ADF	31.14	30.22	37.71	20.74	30.80	40.18
ADL (Lignin)	6.82	3.37	8.18	6.27	10.64	13.26
Cellulose	24.45	26.85	29.53	14.47	20.16	26.92
Hemicellulose	9.31	7.67	5.34	5.06	7.25	8.22
$\frac{(C+H)}{L}$ **ratio**	4.95	10.24	4.26	6.27	10.64	13.26

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