A Study on Sustainable Waste Disposal in South Africa Using Mechanical Biological Waste Treatment

by

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As the candidate's Supervisor I agree/do not agree to the submission of this thesis.

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DECLARATION

I ....Mark Richard Griffith.......... declare that

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DEDICATION

For

Mom and Dad

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ABSTRACT

The landfilling of Municipal Solid Waste poses a threat to the environment in the form of landfill emissions. These emissions are a result of the biochemical breakdown of the waste in the anaerobic landfill environment. A solution to this problem has been found in the form of the mechanical-biological treatment of waste. This technology involves mechanical and biological processing of the waste before it is placed in the landfill. The pretreatment accelerates the degradation of the waste resulting in the landfilling of a more biologically stable product, resulting in a reduction of the emission potential of the landfill.

This research aims at investigating the applicability and efficiency of a passively ventilated MBT windrow system under a sub-tropical climate. The research was conducted in two stages: the first stage focused on the implementation and analysis of the Mechanical Biological Treatment (MBT) process with aerobic windrows, employing the Dome Aeration Technology (DAT) (Mollekopf et al. 2002). Three DAT windrows were constructed at the Bisasar Road Landfill in Durban in order to study the efficiency of the process after different composting timeframes (8 and 20 weeks). The study proved that the use of the DAT technology is a viable option.

The second stage was the analysis of this treated waste in an anaerobic environment, in order to simulate landfill conditions and, thus gain insight into the effect of MBP on landfill emissions. Six lysimeters and 5 columns as well as numerous eluate tests were conducted in order to study the "post-landfilled" behaviour of the waste and the effect that waste treatment, composting time and screening have on liquid and gaseous emissions. A basic cost estimate using the Clean Development Mechanism for financial assistance was conducted. The results of this research were then utilised to make recommendations on sustainable waste disposal options. The findings of the research were that although the MBT did not reduce emission levels sufficiently to allow for a 40 year landfill aftercare period, the benefit over the landfilling of untreated waste is significant.

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BOD_5	5 day Biochemical Oxygen Demand
BOD	Biochemical Oxygen Demand
CDM	Clean Development Mechanism
CERs	Certified Emission Reductions
CH_4	Methane
CN	Condition Number
CO_2	Carbon Dioxide
COD	Chemical Oxygen Demand
DAT	Dome Aeration Technology
H_2	Hydrogen
LFG	Landfill Gas
L/S	Liquid to Solid Ratio
MC	Moisture Content
MBP	Mechanical Biological Pretreatment
MBT	Mechanical Biological Treatment
MBWT	Mechanical Biologcial Waste Treatment
MSW	Municipal Solid Waste
MRF	Materials Recycling Facility
MSOR	Municipal Sorted Organic Residue
N_2	Nitrogen
NO_x	Nitrogen Oxides NO_2 and NO_3
NH_4	Ammoniacal Nitrogen
N_{org}	Organic Nitrogen
N_{humic}	Humic Nitrogen
O_2	Oxygen
RMSW	Residual Municipal Solid Waste
SD	Standard Deviation
TASi	Technical Instructions on Waste from Human Settlements
TKN	Total Kjeldahl Nitrogen
TOC	Total Organic Carbon
TS	Total Solids
VS	Volatile Solids
UNFCCC	United Nations Framework Convention on Climate Change

CHAPTER 1

INTRODUCTION

Sustainability – the first and last word of this thesis, represents a concept that is becoming more and more central to the planning and development of our societies. An increasing awareness of the importance of the general health of our natural environment and recognition that there are many human practices which are detrimental to it, has conceived the concept of sustainability. The fundamental philosophy behind this concept is the implementation of activities that the environment is able to sustain without negative long-term impacts. The term "appropriate" should fit the answer to the question of sustainability in the context of a developing country which has many other needs to meet. This research is aimed at finding/proposing methods in order to solve the environmental problems associated with landfilling large volumes of untreated waste – methods that are both appropriate and sustainable.

Modern society has been challenged with the problem of how to properly manage the ever growing volumes of waste that is generated, and how to eliminate the harmful impacts that the high concentrations of waste pose to both human health and that of the environment as a whole. Landfills are one such solution that has been implemented is whereby waste is collected and contained in large, concentrated volumes. Developments in the understanding of waste have revealed that although the landfill does solve problems, it creates additional long-term environmental problems (Robinson, 1996). This has motivated research into the development of technologies that reduce the impacts associated with landfilling. One option is the Mechanical Biological Treatment (MBT) of solid waste (Heerenklage, 1995; Soyez, 2002; Muntoni, 2005). As the majority of the environmental issues that are associated with landfills arise from the emissions produced during the anaerobic digestion of the organic matter, MBT focuses on accelerating the stabilization of the organic matter though optimised biological degradation, before the waste is landfilled (Soyez, 2002).

The MBT technology options range from sophisticated in-vessel systems that utilise forced aeration and mechanical turning to simpler open windrow systems which rely on passive ventilation through the chimney effect (Mollekopf et. al., 2002; Muntoni, 2005)
This research investigates the implementation of a robust open aerobic treatment technology, known as the Dome Aeration Technology (DAT) which uses passive aeration based on the chimney effect (Mollekopf et. al. 2002). As there are no active

MBT operations in South Africa, three trial windrows using the DAT were constructed and studied at the Bisasar Road Landfill site in Durban. Waste from these windrows was characterised and placed in anaerobic leaching columns and lysimeters in order to evaluate the emissions of waste after varying stages of treatment. These column and lysimeter tests were used to gauge the benefit of waste treatment and to predict the pollutant potential of the treated wastes after landfilling. A financial evaluation of the cost implication for the implementation of a local MBT operation using the DAT system was developed in order to complete the feasibility assessment of a full scale DAT waste treatment operation in Durban, South Africa.

The objectives of the research were:

- To assess the viability of implementing the DAT under the sub-tropical climate of Durban and to offer recommendations for future local DAT operations;
- To investigate the emissions of treated waste once landfilled and to assess the benefit gained through waste treatment.
- To investigate the effect of screening the treated product on the emissions produced during anaerobic degradation of the treated waste.
- To model the pollutant release from the waste in order to assess the polluting potential of the treated waste, and the benefit of MBT over untreated waste.
- To use the findings of the study to recommend a waste disposal operation for the EThekwini Metro.
- The assessment of the Mechanical Biological Treatment of Waste as a sustainable waste management solution

The research and the findings are covered in the following nine chapters of this dissertation. Chapter 2 is a literature review of the current status of waste disposal by landfill, the techniques adopted by South African and European legislations, the sustainability of these techniques and the concept of mechanical biological waste treatment. Chapter 2 also covers the physical and biochemical characteristics of treated waste, discusses the costs of MBT operations, describes the development of a Materials Recycling Facility (MRF) at the Mariannhill Landfill Site and introduces the Clean Development Mechanism (CDM) concept as a means to assist in the funding of MBT projects. Chapter 3 focuses on the natural ventilation system of the Dome Aeration Technology and reviews the implementation of the DAT at the Cottbus landfill site in Germany. Chapter 4 covers the field study of three DAT windrows constructed in Durban using standard landfilling equipment, the results of which are presented in

Chapter 5, along with the recommendations that arose from the investigation. Chapter 6 introduces the materials and methods of characterising the waste through eluate tests, columns and lysimeters as a means to evaluate the landfilled behaviour of waste. In Chapter 7 the results from the characterisation and column and lysimeter study are presented and the long-term pollution potential of the waste is estimated. Chapter 8 presents the development and results of a leaching model with the aim of predicting the long term release of pollutants from the waste. Chapter 9 presents a recommended waste disposal strategy based on the findings of the DAT and lysimeter studies as well as the financial implications of such and operation that includes an estimation of the potential revenues from Certified Emission Reductions (CERs) as a means to fund the project. The final conclusions and recommendations are presented in Chapter 10, where the objectives listed above are addressed and the results of the research are evaluated.

CHAPTER 2

SOLID WASTE DISPOSAL

2.1 INTRODUCTION

Over 95% of waste in South Africa is disposed in landfills (DWAF, 2005). The waste contained within a landfill is stabilized by anaerobic organisms that metabolise the degradable organic fractions within the waste. During these anaerobic degradation processes, dangerous gases, offensive odours and toxic leachates are emitted from the landfill and present a risk to the environment. This presents a problem to the current means of waste disposal in South Africa.

Stabilisation of the waste prior to landfilling results in the elimination of the risks posed by these landfill emissions. This approach has been adopted in European countries, where an appropriate waste treatment process is required prior to landfill (Heerenklage et al., 1995; Bone et. al., 2003). Biological treatment of the waste stabilizes the degradable organic fraction of the waste through enhanced biological degradation. The biochemical processes are optimised and the stabilisation process is achieved far more rapidly than for conventional anaerobic landfills (Heerenklage et al., 1995). On the other hand, thermal incineration oxidizes the waste material and produces a stable residue as well as allowing for the generation of energy during the combustion of the fractions with a high calorific value (Williams, 1998). These practices are not applied to solid waste treatment in South Africa where legislation allows for the landfilling of organic-rich, untreated waste, resulting in the formation of landfills that present a long-term environmental risk (Robinson, 1995).

This chapter serves to describe the processes of waste decomposition and the problems associated with landfill emissions. The potential of waste treatment, and in particular Mechanical Biological Treatment, as solution to this problem is assessed with the MBT processes and technologies explained. The potential for the implementation of the MBT technology in the South African context is also addressed with the particular focus on low maintenance, low cost and appropriate systems.

2.2 SOLID WASTE GENERATION AND MANAGEMENT

The definition of waste is subjective, as what may be deemed to be "waste" by one individual, may represent a valuable resource to another (Williams, 1998). The South African White Paper on Integrated Pollution and Waste Management provides the following definition of waste (Visser, 2002):

> *"**Waste** – an undesirable or superfluous by-product, emission, or residue of any process or activity which has been discarded, accumulated or been stored for the purpose of discarding or processing. It may be gaseous, liquid or solid or any combination thereof, and may originate from a residential, commercial or industrial area. This definition includes industrial waste water, sewage, radioactive substances, mining, metallurgical and power generation waste."*

Solid waste is categorized according to the threat that it may pose to plant animal or human life. Two categories exist: General Waste and Hazardous Waste (DWAF, 1998).

The focus of this research is on general waste, which includes, but is not limited to, the following fractions: food waste; garden wastes; paper; textiles; plastics; non-ferrous metals; ferrous metals; rubber; dirt (eg. sweepings); construction waste; builder's rubble; glass; ceramics; wood waste (eg. Sawdust) (Peavy, 1985). The vast range of materials that is included in general waste, results in a large degree of heterogeneity. The purpose of solid waste management is the removal of the refuse from the producer and the disposal by means that are stipulated by the local authorities (Williams, 1998). After generation, the waste may pass through several processing stages that extract fractions from the waste stream for the purposes of financial gain or for reasons of enforced environmental protection. These processes are waste re-use, materials recovery and waste treatment. The remaining fraction of the waste stream is then disposed of, principally by landfill (Williams, 1988). The processes are illustrated in Figure 2.1.

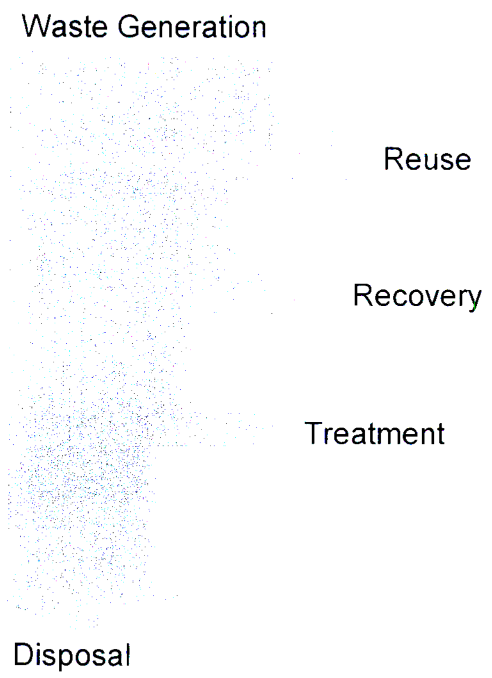

Figure 2.1 Waste Stream Processing Steps (adapted from Williams, 1998)

Reuse - Objects are removed and processed for re-use. Examples include re-treaded tyres and refillable drinking bottles (Williams, 1998).

Recovery - Energy is recovered either by processing of selected wastes for use as fuel, combustion of waste or combustion of methane that is produced by anaerobic degradation of the putricible content of the waste. Materials are recovered, either by the recycling of useful fractions or by the generation of compost from the organic fractions (Peavy, 1995; Williams, 1998).

Treatment - The waste is treated in order to comply with disposal standards. The aim of waste treatment is the removal of waste fractions that may pose a threat to the environment. Treatment options include thermal incineration and mechanical biological treatment (Muntoni, 2005).

Disposal - Disposal is the final option for the management of materials where the application of the preceding processing strategies incurs environmental and economical costs that outweigh the benefits (Williams, 1998).

The application of these processing stages may negate the need for further processing. For example thermal combustion for energy recovery is equivalent to the incineration treatment (Williams, 1998). In South Africa there is no legislation that demands the processing of general waste prior to landfill disposal, and as a result the only "waste treatment" processes that are applied are voluntary re-use and recycling. The environmental benefit of waste treatment prior to disposal is yet to be realised in this country. The purpose of this research is to evaluate biological waste treatment in South Africa as a viable option.

2.3 BIOLOGICAL STABILISATION OF WASTE

Biological waste treatment processes are effectively engineered microbial culture systems, which convert the large amounts of organic material into more stable forms (with the exception of methane). The metabolism of degradable compounds includes the catabolic reactions through which organisms extract energy by oxidative degradation, and the anabolic reactions, which involve the synthesis of cellular protoplasm (Grant and Long, 1981).

There are two main pathways that qualify the catabolic processes of oxidative degradation: anaerobic fermentation, where the oxidation occurs in the absence of any added electron acceptors, and respiration, where oxidation occurs in the presence of a terminal electron acceptor (Pelczar et. al. 1977). If oxygen (O_2) is present, the respiration is aerobic. Aerobic respiration utilises oxygen as the ultimate electron acceptor, while anaerobic respiration, which occurs in the absence of oxygen, utilises molecules other than oxygen as the ultimate electron acceptor (Bailey et. al., 1977). Besides the strictly aerobic and anaerobic organisms, facultative anaerobic organisms are able to utilise oxygen when it is present, and other molecules in its absence. Through these biological reactions, molecules of high energy value (enthalpy) are transformed into molecules of lower energy value which are more stable (Bailey et. al., 1977).

2.3.1 Aerobic Degradation

For most aerobic microorganisms, water (H_2O) and carbon dioxide (CO_2) are the metabolic end products of the biological breakdown of organic compounds (Bailey et. al., 1977). The extraction of free energy from the organic compounds is relatively high, and may be greater than 60% for the breakdown of glucose (Bailey et. al., 1977). The remaining energy would be released as heat. The processes of aerobic degradation in the context of composting are discussed in more detail in section 2.8.

2.3.2 Anaerobic Degradation

The end products of aerobic respiration have lower utilizable free energy than the products of anaerobic respiration. The anaerobic end products from one microorganism may be metabolised by other microorganisms. As a result the transformation of waste under anaerobic conditions, from complex macromolecules into biogas (methane (CH_4), carbon dioxide and other gases), involves the activity of several groups of microorganisms.

Four different phases can be distinguished in the anaerobic biochemical reactions- Hydrolysis, Acidogenesis, Acetogenesis and Methanogenesis (van Haandel et. al, 1994, Christensen et. al., 1996).

Hydrolysis

During this extra cellular process, complex particulate matter is converted into dissolved compounds, which have a lower molecular weight. This allows for the absorption of compounds across the cell membrane. The products of this phase include amino-acids, carbohydrates, simple sugars, fatty acids and other low-weight organic compounds (Farquhar and Rovers, 1973; van Haandel et al, 1994).

Acidogenesis

The acidogenic stage is a fermentation process where the microorganisms convert the dissolved compounds generated during the hydrolysis step into simple organic compounds such as volatile fatty acids, alcohols, lactic acid and mineral compounds such as carbon dioxide, hydrogen (H_2), ammonia (NH_3), and hydrogen sulphide (H_2S). Although most of the bacteria that carry out the acidogenic fermentation are anaerobes, there are some that are facultative and can metabolise matter through oxidative paths (van Haandel et. al., 1994).

Acetogenesis

The products of acidogenesis are converted into acetate (CH_3COOH), hydrogen and carbon dioxide. Due to the complex mix of matter found in the waste, both carbon dioxide and hydrogen are produced during the formation of acetic acid. Generally more hydrogen than carbon dioxide is formed (van Haandel et al, 1994).

Methanogenesis

Methane is produced from via two routes: Acetotrophic Methanogenesis and Hydrogenotrophic Methanogenesis. During acetotrophic methanogenesis, acetate is converted into methane (2.1), whilst during hydrogenotrophic methanogenesis carbon dioxide is reduced by hydrogen to form methane (2.2). The stochiometry for the two reactions is as follows (Metcalf et al, 1991).

$$CH_3COOH \rightarrow CH_4 + CO_2 \qquad 2.1$$

$$4H_2 + CO_2 \rightarrow CH_4 + 2H_2O \qquad 2.2$$

The methanogenic bacteria, known as "methanogens" or "methane formers" are strict anaerobes and also very sensitive to environmental conditions of temperature and pH (Hammer et. al., 2001). The hydrogenotrophic bacteria grow faster than the acetotrophic bacteria, consequently the conversion of acetate to methane is rate limiting.

In waste management, aerobic biological activity is utilised in waste stabilization processes, while anaerobic activity is encountered waste stabilization operations and also occurs naturally in landfills.

2.4 WASTE DISPOSAL BY LANDFILL

The term Landfill describes an area that is used for waste disposal on land. Globally, landfilling is the most widely used means of ultimate waste disposal and in most countries of the world landfills are the only disposal option, receiving more than 85% of waste produced (Stegmann, 1995). In South Africa the rate is higher with over 95% of all urban waste is disposed of in landfills (DWAF, 2005). Although there are other forms of waste treatment, the landfill is still required as a final disposal option for the residues (Peavy, 1985; Williams, 1998). The modern sanitary landfill employs daily cover, engineered barrier systems and gas management in order to reduce the risk that the waste poses to the environment.

The waste disposed in landfills typically contains a significant proportion of putrescible organic substances. The layers of compacted waste are isolated from the atmosphere, resulting in the formation of anaerobic conditions within the landfill. Populations of microorganisms become established and metabolise the degradable fractions of the waste, resulting in the slow progressive decomposition of the organic material

(Gendebien et al, 1992). In essence, the landfill is a large anaerobic digester. The process of waste degradation produces harmful gases that carry offensive odours and the moisture that percolates through the waste body becomes highly polluted.

2.4.1 Landfill Emissions

The greatest environmental threats that landfills pose are the gaseous by-products of the degradation processes (landfill gas), and the polluted leachate that percolates through the layers of waste.

Landfill Leachate

Landfill Leachate is the liquid that has percolated through the waste body, and contains dissolved or suspended material from the refuse (Peavy et al, 1985). The nature of the leachate depends on the type and composition of the waste that it comes into contact with, the state of waste biodegradation, the moisture content of the waste body and the landfill operational procedures (Williams, 1998). Leachate originates from the natural moisture in the waste, rainwater and other liquid input – e.g. co-disposal sites.

Landfill leachate is highly polluted containing high concentrations of degradable organic compounds and nutrients, solids, large bacterial populations, pathogens, toxins and heavy metals (Table 2.1). These elements, as well as the low pH encountered in the Acid Phase are among the most significant of the environmental problems associated with landfill leachate and may individually, or cumulatively, contribute to the damage of natural water systems, or adversely affect human health (Peavy, 1985; Williams, 1998, Olufsen, 2003).

Landfill Gas

Landfill Gas (LFG), also referred to as biogas, is generated by bacteria during the anaerobic degradation of waste. Landfill gas is essentially a combination of gases and volatile compounds, primarily carbon dioxide and methane. Other gases such as hydrogen and hydrogen sulphide are also produced, and the gas may contain numerous trace compounds. The predominant gases present in LFG depend on the stage of biodegradation, the moisture content of the waste, and the nature of the waste. The volume of gas produced varies according to different waste streams and ranges from 60-400 Nm^3/t (Bowers, 2002). The major constituents of LFG are presented in Table 2.2

The gaseous emissions from landfills pose problems to the vegetation, human population and groundwater on or surrounding the landfill site, and because methane has a global warming factor of 20-25, the gas also contributes to global climate impacts (Gendenbien et. al., 1992; Bowers, 2002).

Table 2.1 Typical Range of Landfill Leachate Composition and South African General Discharge Standards (Adapted from Qasim et. al., 1994) and Strachan

Parameter	Unit	Landfill Leachate	General Discharge Standard
pH		5.1 - 8.5	5.5 - 9.5
Conductivity	mS/m	38 - 5 200	250
Total organic carbon (TOC)	mg/l	80 – 20 000	-
Chemical oxygen demand (COD)		150 - 152 000	75
5 day biochemical oxygen demand (BOD_5)		100 - 90 000	-
Fatty acids (as carbon)		1 – 22 500	-
Total suspended solids		100 – 2 000	-
Total Kjeldhal nitrogen (TKN)		50 – 5 000	-
Organic nitrogen (N_{org})		1 – 2 000	-
Ammoniacal nitrogen (NH_4-N)		1 – 4 110	10
Nitrate nitrogen (NO_3-N)		0.1 – 50	-
Nitrite nitrogen (NO_2-N)		0 – 25	-
Alkalinity as $CaCO_3$		300 – 16 000	250
Calcium		100 – 3 000	
Magnesium		25 – 1 150	-
Potassium		50 – 4 000	-
Sodium		10 – 3 100	-
Chloride		30 – 5 000	-
Sulfate		0 – 1 600	-
Total Iron		0.4 – 2 300	0.3

2.4.2 Landfill Emission Stages

The characteristics of significant landfill emissions can be classified according to three stages, the Aerobic Stage, the Acid Stage and the Methanogenic Stage (below) as described by Robinson (1989).

found to range from months to years. For warmer climates such as in subtropical areas of South Africa and Australia, the time frame is six to nine months (Lombard, 2000; Bowers, 2002).

The gradual establishment of the sensitive methanogenic bacteria results in the formation of a dynamic equilibrium between the acid phase bacteria and methanogenic bacteria. This signifies the end of the acid stage and the beginning of the methanogenic stage (Robinson, 1989).

Methanogenic Stage

The strictly anaerobic methanogenic microbes convert the soluble organic compounds into carbon dioxide and methane. It is soluble organic compounds that are largely responsible for the low pH, high COD levels and high BOD to COD ratio of the leachate produced during the acid stage (See Table 2.1). As a result of the removal of these compounds, the leachates produced during the methanogenic stage are significantly less aggressive, with low COD values, low BOD to COD ratio and a more neutral pH. The ammonical nitrogen concentrations, however, remain high. The timescale for the reduction of leachate levels to environmentally acceptable levels has been estimated to be 300-1000 years (Muntoni, 2005).

The gas production rate, an indication of biological activity, reaches a peak for several years, and gradually decreases until the waste is fully stabilised, and there is no longer material available for the sustenance of the bacteria (Robinson, 1989; Rohrs et. al, 1998; Lombard, 2000).

2.5 SOUTH AFRICAN WASTE MANAGEMENT LEGISLATION

Driven by the need for an environmentally acceptable, yet cost effective means of waste disposal, the South African government released a series of documents called the Minimum Requirements. The first series was released in 1994, a second edition was published in 1998 and the third edition published in 2005. The three objectives of the Minimum Requirements for Waste Disposal by Landfill (DWAF, 2005) are:

- To improve the standard of waste disposal in South Africa.
- To provide guidelines for environmentally acceptable waste disposal for a spectrum of landfill sizes and types.

- To provide a framework of minimum waste disposal standards within which to work and upon which to build.

The practices used to achieve the abovementioned objectives must be the Best Available Technology Not Entailing Excessive Cost (BATNEEC) (DWAF, 1998). In order to meet this requirement of practical and affordable environmental protection, a series of graded standards are applied to different classes of landfill. The landfill classification takes into account the type of waste disposed of, weather it be general of hazardous waste, size of the waste stream, and the potential for the generation of significant quantities of leachate. The current approach of land disposal is "Concentrate and Contain" (Bredenhann et. al., 2003), with the objective of elimination of leachate generation through the installation of impermeable lining systems and final capping layers.

2.6 SUSTAINABLE WASTE DISPOSAL SOLUTIONS

The concept of sustainability may lead to problems associated with expectations that arise from different interpretations of the term. As Röhrs and Fourie (2002) describe "'Sustainable Landfilling' is a concept often alluded to, but seldom defined". It is thus important to establish a definition of 'sustainable' that sets the context through which the following topics are viewed. The basic concept of sustainability implies that the environmental concern does not last for a timeframe that extends past the length of one generation. There are other concerns about the standards and levels of understanding that are used to evaluate the environmental risks (Robinson, 2000). However, for the purposes of working towards a solution, the following definition of sustainable operations will be used:

A sustainable operation is one that requires no resource input and presents an acceptable risk to the environment, according to contemporary standards, after a period extending no longer than forty years from the decommissioning of the project

An important consideration in the abovementioned definition is the term "acceptable risk". For the purposes of this research, the legislative discharge standards will be utilised. Furthermore, the costs incurred by a waste management system must lie within the financial means of the community that it serves. This highlights the importance of the concept of *Appropriate Technology*.

2.6.1 The Sustainability of Landfilling Waste

Waste degradation processes are reliant on a supply of moisture thus the installation of landfill capping layers which prevent the ingress of water, referred to as encapsulation, retards the stabilization of waste (Knox, 2000).

The timescales involved for the complete stabilization are very long (Robinson, 1996), and over the last two decades the estimates of the timescale for the potential polluting life of a large modern landfill has increased from early estimates of 20-40 years to greater than 500-1000 years (Hall et al., 2006). In light of the abovementioned definition of sustainability, it is clear that the process of landfilling of highly degradable waste and preventing moisture ingress, as per the guidelines in the minimum requirements (DWAF, 2005) is an unsustainable practice. It is necessary to develop and introduce appropriate techniques to accelerate the stabilization processes. This need has been realised by the European Union (EU), and has resulted in the implementation of legislation controlling the nature of material that may be landfilled.

European Legislation

The European Council Directive on the Landfilling of Wastes requires member states to ensure that the amount of biodegradable material that is landfilled is progressively reduced over a 15 year time period, to 35% of the total amount produced in 1995. The directive also requires that waste can only be landfilled if it is subjected to treatment or incineration (Bone et. al., 2003).

In 1993 the German government implemented the "Technical Instructions on Waste from Human Settlements" (TASi), a regulation that controls the standards of landfilling operations, also listing allocation values for the deposited waste (Soyez et. al., 1997). A significant criterion that must be achieved is a threshold value for the organic content of the landfilled material which must fall below the level of 5%, as determined by ignition loss, or 3%, determined as Total Organic Carbon (TOC), thus necessitating the pretreatment of waste before landfilling. A similar standard has been implemented by Austria, which limits the ignition loss and TOC which must fall below the respective values of 8 and 5% in order for the material to be landfilled (Raninger et. al. 1997).

2.6.2 Treatment of Waste

Waste treatment is a process applied to waste before it is landfilled. The objective of the process is the accelerated stabilisation of the degradable component of general waste, and therefore elimination of the harmful emissions that result from the anaerobic degradation that occurs in landfills. The two most commonly implemented options for general waste treatment are thermal (incineration) and biological treatment. The biological treatment can be applied as a single treatment operation, or in combination with thermal treatment (Soyez, et. al., 1997). Once the waste has been processed, the residues may be landfilled.

Waste Incineration

Incineration involves the oxidation of the waste material at high temperatures (750-1000°C) which results in the stabilisation of organic and/or hazardous compounds. There are many advantages of thermal treatment (Williams, 1998):

- The product of incinerator plants is biologically sterile and the approximate volume and mass reduction for municipal solid waste (MSW) is 90% and 65% respectively.
- The combustion of waste can be used as a means of energy recovery; an example being the utilisation of the thermal energy for co-generation (electricity and heating).
- The bottom ash residues can be recovered and used, for example, as secondary aggregates in construction.
- Incineration is also the best practical environmental option for the treatment of many hazardous wastes.

However, there are also disadvantages of incineration, one being the higher running costs and large capital investment required for the plant construction (Muntoni, 2005). The high capital cost of incinerator plants also implies that the plant design must be tied to long-term waste disposal contracts, resulting in little flexibility in terms of changing disposal trends. An example of this is the potential increased removal of materials of high calorific value, such as paper or plastics, which may adversely affect the incinerator performance, if the design did not allow for such activities (Williams, 1998). Although modern incinerators do comply with existing emission regulations, public concern of the adverse effects of incinerator emissions remains high and public concern, particularly of the release of toxins from the flue gases, may cause strong social opposition to the development of thermal treatment plants in populous areas Williams, 1998).

Mechanical Biological Treatment of solid waste

The Mechanical Biological Treatment of Solid waste - also known as Mechanical Biological Pretreatment (MBP) or Mechanical Biological Waste Treatment (MBWT) - involves the mechanical preparation of the waste, followed by a period of optimised biological degradation. The biological stabilisation of the waste is achieved far quicker through MBT than in a landfill with studies showing that a period of 8-24 weeks will produce a biologically stable product, depending on the technology employed (Brinkmann et. al, 1995; Leikam, et. al, 1997; Mollekopf, 2002; Soyez, 2002; Bone, et. al, 2003).

The advantages of MBT over waste incineration are reduced gaseous emissions and lower operational costs (Leikam et. al., 1997; Soyez et. al. 1997). Leikam et. al. (1997) state that in order to ensure an economically viable incineration plant, the waste throughput should be at least 150 000 to 200 000 tons/annum. Many communities do not produce these quantities of waste, and thus large transportation costs would be incurred for the transportation of waste to regional thermal treatment facilities. The disadvantage of MBT is that the mass, volume and emission potential of the residues is higher than that of thermal treatment processes (Leikam, et al. 1997).

The higher emission potential has proved to be a great shortcoming for MBT. Although most of the TASi values were met by MBT, the organic content of the waste and leachate - the ignition loss and the TOC value of the waste and leachates - could not be met by a technologically viable period of processing (Leikam et. al., 1997; Soyez et. al., 1997). Waste scientists have disputed the use of ignition loss as a parameter for the evaluation of MBT waste. It is a well-established fact that the ignition loss is not fully representative of the activity inside a landfill. The ignition loss includes the whole organic content, of which there is a portion, e.g. plastic, that is not available for biological degradation in the landfill. Another potential oversight in the regulations is the possible positive effects of humic substances that have a high capacity to bind heavy metals or other toxic organics (Soyez et. al., 1997). After a lengthy period of political discussions and scientific examinations, waste with a TOC content of less than 18% or 250mg/g dry mass (eluate), may be landfilled (Mollekopf et. al., 2002). Additional regulations with regard to the gaseous and liquid emissions from MBT facilities were added. The new regulations require a closed composting step, during which all exhaust gases must be collected and purified (Mollekopf et. al., 2002). Thus the relaxation of the TOC parameters which has allowed for MBT to be utilised has adversely affected the financial attractiveness of the process through the exclusion of the less expensive open systems.

2.7 AN OVERVIEW OF MECHANICAL BIOLOGICAL TREATMENT

The MBT process fundamentally encompasses two activities - the mechanical processing of the waste and a period of biological degradation.

2.7.1 Mechanical Treatment

The mechanical treatment step removes useful fractions from the wastes stream and prepares the biodegradable portion for optimal microbial activity. The operation involves sorting, shredding/homogenisation and adjustment of biological parameters.

Sorting

Screening or sieving the waste, using a screen size of between 60 and 100mm has proven to be effective in sorting the waste that is highly biodegradable from those wastes with a high calorific value. A grain size distribution of un-shredded residual municipal solid waste (RMSW) using a mesh size of 80mm shows that over 90% of the vegetable residues passes through the sieve, while 70% of the fractions high in calorific value (e.g. paper and plastics) remains in the sieve overflow. Although material recovery is possible through sieving, in general the material is not suitable for recycling due to contamination by the vegetative residues and a more viable option is usage as refuse derived fuel if the process is used in conjunction with thermal treatment plants (Leikam et. al., 1997). The removal of interfering materials (e.g. large stones) protects the downstream processing equipment. The removal of ferrous materials has several benefits: the volume of waste to be treated further is reduced, the metals removed can be recycled and the quantity of heavy metals, which are an environmental hazard, is reduced. The sorting process can be applied both before and after the biological treatment step (Soyez, 2002).

The following processes are available for the sorting of the waste:

- rotating sieving drums
- hand sorting
- magnetic separation (ferrous materials)
- ballistic separation (light fraction)

The recent establishment of the material recycling facility at the Mariannhill Landfill site in Durban is an exciting development in the field of waste management in South Africa and is discussed in more detail in Section 2.7.2.

Shredding

Shredding the waste has the benefit of volume reduction and increasing the surface area of the waste, allowing for enhanced biological activity. Common means of waste shredding are:

- high speed hammer/impact mills
- low speed knife shredders
- screw, worm and cascade mills.

Shredding is applied before the biological treatment step.

Biological Parameter Adjustment

In order to optimise the biological degradation, the physical and chemical nature of the waste should be controlled. The three major parameters are waste moisture content, particle size and carbon/nitrogen ratio. These parameters are discussed in more detail in Section 2.8.

2.7.2 Commercial Materials Recycling Facility at Mariannhill Landfill Site

The commissioning in September 2007 of the privately owned and operated materials recycling facility at the Mariaanhill Landfill site represents a significant step towards sustainable waste management in South Africa. The facility, located on site, is completely self sufficient, requiring no funding from the landfill operations. The MRF operation is comprised of the following elements (Purchase, 2008):

- Infeed Conveyor with Bag Breaker (Plate 2.1)
- Presort Line
- Trommel (Plate 2.2)
- Main Picking Line
- Residue Picking Line

The significance of this operation in the context of MBT is that the sorting causes the opening and homogenisation of the unsorted MSW that arrives at the site and provides a "free" mechanical processing step in the MBT operation. The wetting of the waste and mixing in of bulky material may be the only additional steps required. Furthermore, the removal of non-degradable recyclable materials from the waste stream would result in a more efficient biological treatment process on a mass/mass basis.

Plate 2.1 Mariannhill MRF Bag Breaker and Infeed Conveyor (Source, RE)

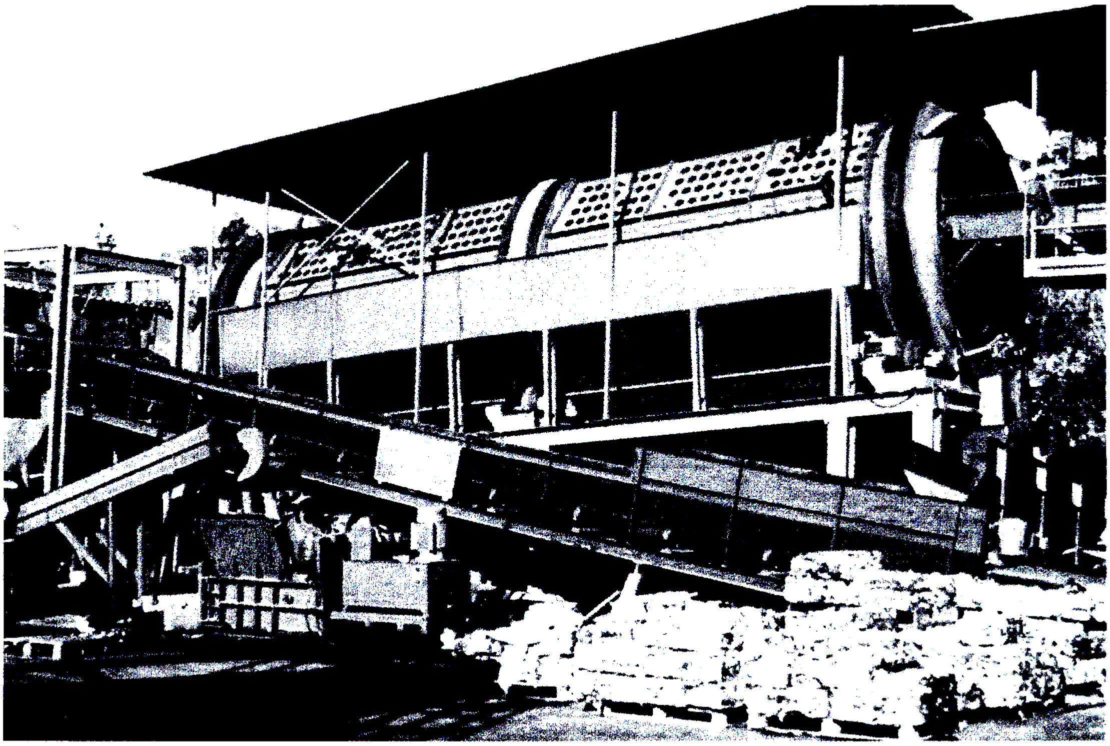

Plate 2.2 Mariannhill MRF Trommel (Source, RE)

Unfortunately, this plant was only commissioned after the completion of the waste treatment stage of this research, presented in Chapters 4 and 5 and thus the treatment of the MRF residue did not form part of this study. However, it should be considered for future research into waste treatment.

2.7.3 Biological Treatment

The biological treatment step serves to reduce the quantity of biodegradable matter in the waste; it can either be anaerobic-aerobic or purely aerobic (Soyez, 2002).

Anaerobic Treatment

Anaerobic treatment has the advantages of smaller space requirements, minimal odour problems and a net gain in energy if the biogas produced during the biodegradation processes is collected and used. However, the anaerobic treatment must be followed by an aerobic step due to the inefficient degradation of certain materials such as lignin under anaerobic conditions, and the need for the treatment of high concentrations of ammoniacal-nitrogen that develop under anaerobic conditions (Krogmann, 1995; Soyez, 2002).

A modern development in waste treatment is the anaerobic treatment of the waste for the recovery of biogas for energy, after which the waste is centrifuged and combusted in thermal treatment plants, negating the need for an aerobic treatment step (McKendry, 2008).

Aerobic Treatment

Aerobic treatment can be used as a one step biological treatment and requires a continuous supply of oxygen for the microorganisms. Numerous processes are available for the provision of oxygen, and the three major aeration principles that are applied are waste agitation (turning), forced aeration and natural (convective/diffusive) aeration (Grey, et al, 1971). The aerobic process does not produce methane thus energy generation from biogas is not possible, however a large amount of heat energy is produced. This heat energy serves to further enhance the degradation processes, and can also be utilised for passive aeration of the material (Krogmann, 1995).

Aerobic treatments are the most robust, economical and widely utilised technology and will only be considered for this study. The fundamentals behind aerobic treatment can be based on the principals of composting operations.

2.8 AEROBIC DEGRADATION

The biochemical breakdown of the degradable organic material under aerobic conditions that occurs during aerobic pretreatment, also known as composting, is a dynamic process that occurs within a pile of degradable material and consists of the combined activities of a succession of varied microbial populations, each suited to an environment of limited duration, and each active in the decomposition of particular groups of organic materials (Gray et al, 1971). The process may be described according to the progression of four stages, namely the mesophilic, thermophilic, cooling and maturing stages (See Figure 2.2). The products of the biological reactions are heat, carbon dioxide, water, cellular protoplasm and humus.

Mesophilic stage

At the start of the process, the mass is at ambient temperature, and is usually slightly acidic. The indigenous mesophilic organisms multiply causing the temperature to rise rapidly. The products of this stage include simple organic acids, causing the drop in pH to acidic ranges. As the temperature exceeds 40°C, the activity of the mesophilic organisms is reduced, and the degradation is taken over by the thermophiles, starting the thermophillic stage.

Thermophillic stage

At the onset of the thermophillic stage, the pH rises, and ammonia may be liberated if excess nitrogen is present. The thermophilic fungi die off at temperatures in excess of 60°C, but the spore-forming bacteria and actinomycetes continue the degradation. At these high temperatures cellulose and lignin are scarcely affected, but waxes, proteins and hemicelluloses are readily degraded. As the readily degradable fractions are consumed, the reaction rate decreases, and as the rate of heat generation falls, the mass starts to cool, ending the thermophillic stage, and starting the cooling stage.

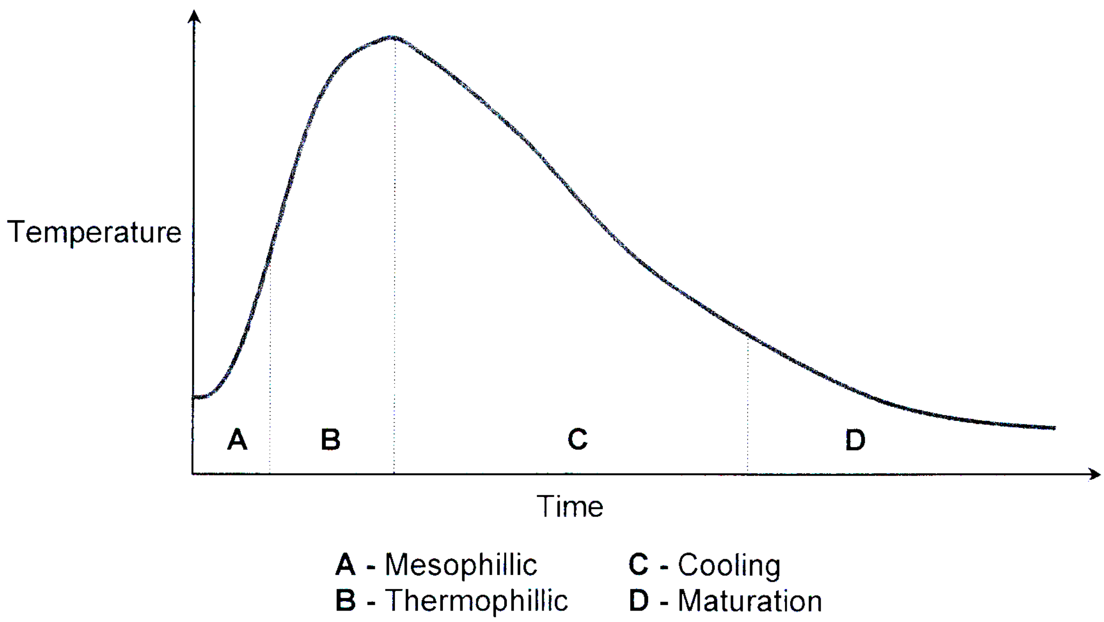

Figure 2.2 Temperature variation with time, indicating the stages of microbial activity in a compost heap (Adapted from Gray et. al., 1971).

Cooling stage

As the temperatures once again become favourable for certain species of microbes, these organisms may become re-established from heat resistant spores, or from invasion from the outer cooler areas. At temperatures below 60°C the thermophilic fungi become re-established, and continue the degradation of the cellulose fractions. The temperatures continue to drop toward ambient. At 40°C the mesophilic organisms once again become active, starting the maturing stage.

Maturation stage

During the maturation stage there is little heat evolution and weight loss as complex reactions give rise to the stable end product, humus. The first three stages of the cycle take place relatively quickly, and are over in a matter of days to weeks, while the fourth stage of maturation usually requires a period of months (Gray et al, 1971).

2.8.1 Aerobic Biodegradation Factors

The rate of biodegrading is dependant on numerous factors and thus optimisation of these parameters is desirable in the design and operation of aerobic treatment systems.

Oxygen

The biological processes are dependant on an adequate supply of oxygen if they are to remain aerobic. The oxygen requirements are dependant on the type of material (e.g. nutrients, particle size), the process temperature, the stage of the process (e.g. higher oxygen demand in the early stages) and the process conditions (e.g. moisture content, pore structure) (Stentiford, 1996). If the processes lack oxygen, anaerobic conditions can develop and the rate of degradation will decrease and also result in the generation of offensive odours and methane (EPA, 1995). The pile should have enough void space to provide for the free movement of air, and the supply of oxygen, and the removal of carbon dioxide and other gases. Oxygen concentrations of 10-15% are considered adequate for maintaining aerobic conditions, although this value may be as low as 5% for material with a low oxygen demand, such as leaves (EPA, 1995). Although higher concentrations of oxygen will not adversely affect the biochemical processes, this may indicate that the flux of air may be too high, which may lead to other problems, such as excessive heat removal and desiccation.

Carbon-Nitrogen Ratio

Due to the abundance of biodegradable forms of carbon in municipal organics, the presence of carbon is not usually a limiting factor in the aerobic degradation processes. However, the presence of nitrogen may raise some concerns. The ratio of *available* carbon to nitrogen of 30-50:1 is considered ideal (EPA, 1995; Peavy et al, 1985). Higher ratios tend to retard the decomposition process, while lower ratios may result in odour problems caused by the liberation of ammonia gas. Typical carbon (C) to nitrogen (N) ratios are given in Table 2.3

Table 2.3 Typical C:N Ratios for various materials

Material	C:N Ratio	Sources:
Yard Trimmings	20-80:1*	EPA (1995)
Wood Chips	400-700:1*	EPA (1995)
Sawdust	100:1***	Cillie (1971)
Manure	15-20:1*	EPA (1995)
Finished Compost	15-20:1* < 30:1***	EPA (1995) Cillie (1971)
Raw Sewage Sludge	7-12:1**	Williams (1998)
Municipal Waste	40-100:1* 26-45:1** 30-60:1***	EPA (1995) Williams (1998) Cillie (1971)

Moisture

The presence of moisture is essential for the existence of microbial activity within the material. It has been observed that below a moisture content of 30-35% biodegradation is significantly reduced (Stentiford, 1996) and therefore it may be necessary to add water to materials that have lower-than-ideal moisture content. However, the moisture content cannot be too high, as excessive moisture can impede the oxygen transfer to the microorganisms, resulting in the formation of anaerobic areas, and may also lead to the generation of leachate. Moisture content within the range of 50-60% of total weight is considered to be ideal (Peavy et al, 1985; EPA, 1995)

While the microbial processes generate moisture during the degradation processes, moisture is lost to evaporation. The rate of evaporation usually exceeds the moisture generated by the microorganisms, resulting in a net loss of moisture from the compost pile. This requires the addition of moisture during the composting period, or initially providing enough moisture for the entire composting process. Using larger piles, which, relative to smaller piles, have a lower evaporating surface area per unit volume, can decrease moisture loss.

pH

A pH between 6 and 8 is considered optimum for aerobic degradation. The pH affects the nutrients available to the microorganisms, the solubility of heavy metals, and the overall metabolic activity of the microorganisms. The final pH is dependant on the input materials and the operating conditions. Although the pH can be adjusted artificially, the organic materials are generally well buffered and wide fluctuations in pH are unusual (Cillie, 1971; EPA, 1995).

Temperature

The natural aerobic degradation processes that occur within the waste mass generate heat. The material is generally a poor conductor of heat and therefore the dissipation of this thermal energy is limited resulting in elevated temperatures (Williams, 1998). The optimum temperature range for aerobic degradation is dependant on the design requirements. All microorganisms have an optimal temperature range, and different temperatures result in the dominance of different species of microorganisms, and hence variable degradation outcomes. The temperature range for thermophillic microorganisms is preferred for rapid degradation and destruction of pathogens. The effect of different temperature ranges are given in Table 2.4

Table 2.4 The Effect of Temperature Range on Aerobic Degradation (Stentiford,993).

Temperature Range (°C)	**Result**
>55	Max Sanitation
45-55	Max Biodegradation Rate
35-40	Max Biological Diversity

The pathogen destruction, or sanitization, that occurs within a compost heap, is greater than that reached by thermal destruction only, due to the competition between different microorganism species (Gray et al, 1971) and is beneficial if there is to be human contact with the treated waste. The criterion for sanitisation for various countries are shown in Table 2.5.

Table 2.5 Sanitisation Requirements for Composting in Europe (Stentiford, 1996).

Country	Temperature (°C)	Exposure (Days)
Austria	65	6 (or 2 x 3)
Belguim	60	4
Denmark	55	14
France	60	4
Italy	55	3
Netherlands	55	2

Temperature can be regulated by several means. One way is adjusting the size of the pile as larger piles generate and conserve more heat than smaller piles. In forced aeration systems, adjusting the airflow will affect the heat removal from the pile. This can be used with temperature feedback control to ensure the desired temperature ranges are maintained. In agitated systems, the temperature can be controlled to a certain degree by adjusting the frequency of the mechanical agitation (EPA, 1995; Stentiford, 1996).

Particle Size

In aerobic degradation systems the size of the material particles is significant. Smaller particles usually have a larger surface area to mass ratio, allowing for more microbiological activity on the particle surfaces, facilitating rapid decomposition. However, if the particle sizes are all too small, there will not be enough void space to allow for air movement. The material input can be manipulated in order to create an optimum particle size range (EPA, 1995). The degradation processes naturally reduce the particle sizes.

2.8.2 Composting Operations

The systems used for aerobic composting can be classified according to the reactor type, the materials flow and the means of aeration.

Reactor Type

A reactor is an enclosed vessel in which the composting takes place. A non-reactor process uses an open pile for the composting stage. Reactor composting systems include drums, silos, digester bins and tunnels (Figure 2.3). These systems provide the mixing, aeration and moisture input. The major advantage of using reactor systems is the environmental conditions can be carefully controlled to allow for rapid degradation, and if placed in buildings, produce minimal odour and little or no leachate. Reactor systems usually require an additional curing period using an open windrow system. The disadvantages of reactor systems are their complexity and high construction, operation and maintenance costs (Tardy et al, 1996). Open systems utilise windrows exposed to the elements and are typically less costly, but are also generally less efficient. A windrow by definition is a pile of composting material whose length exceeds it height and width (EPA, 1995).

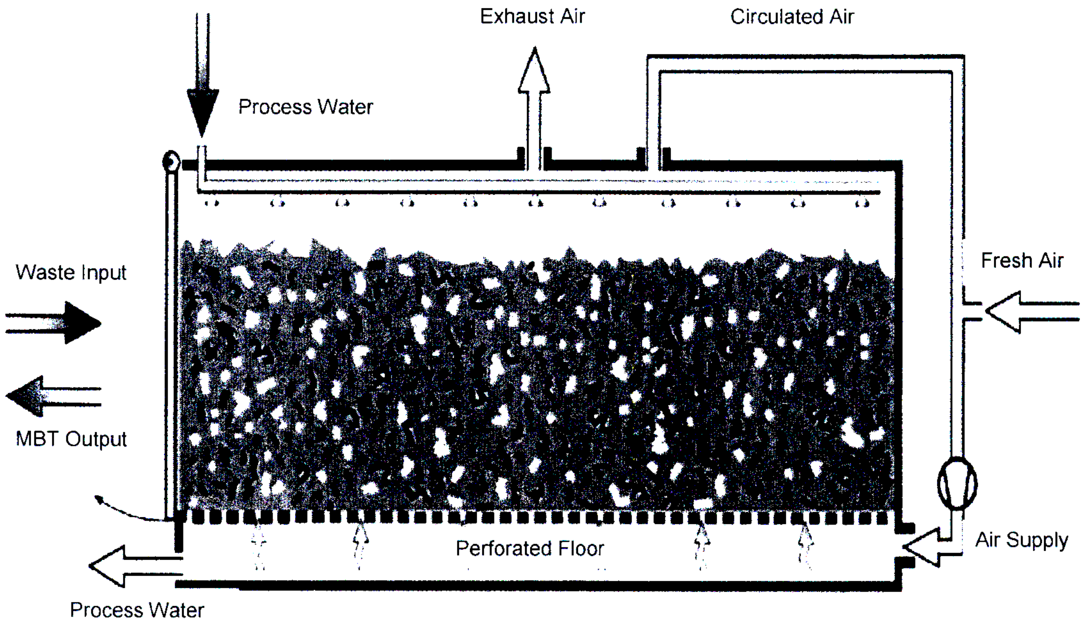

Figure 2.3 Tunnel type MBT reactor (Muntoni, 2005)

Materials Flow

The flow of materials with respect to the composting stage can either be continuous, semi-continuous or batch wise, with retention times ranging from one to four weeks (EPA, 1995; Haug, 1993). In an open windrow operation, the rotting period is relatively long (8 – 24 weeks), and is performed in batches.

Aeration Supply

For all aerobic composting operations, a supply of oxygen is necessary to maintain optimal oxygen levels. Aeration is achieved by agitation, airflow, or a combination of the two. Agitation is where the material is disturbed, exposing the compost to the atmosphere, thereby liberating the compost gas and replenishing the oxygen levels in the void spaces. This method is frequently employed in in-vessel systems, such as rotating drum reactors and may be used in conjunction with forced aeration (Williams, 1998; EPA, 1995).

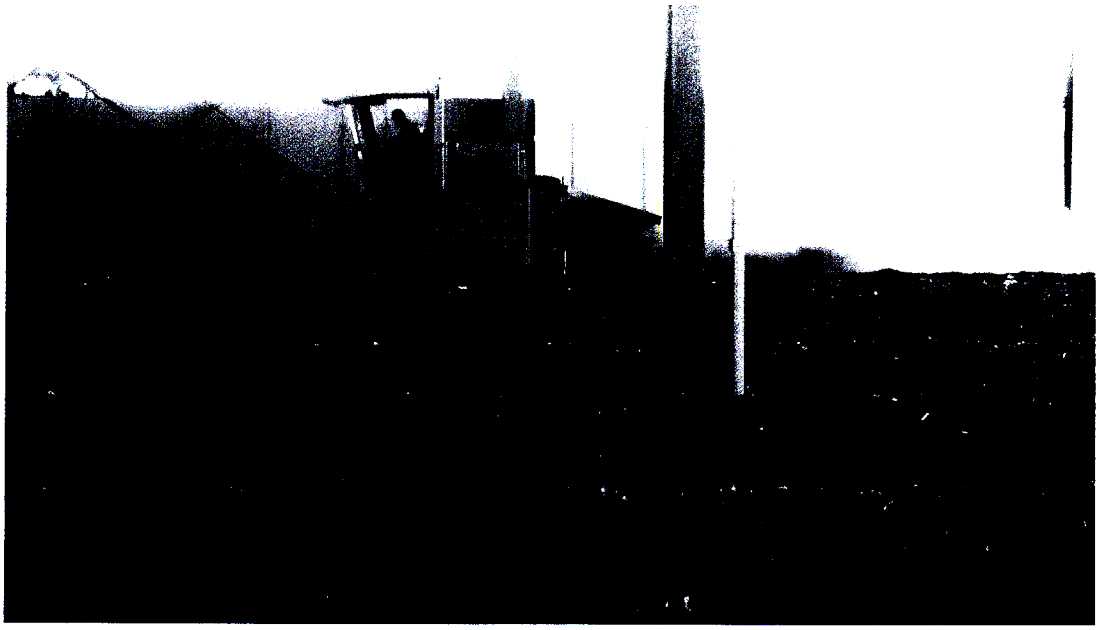

Plate 2.3 A Windrow turner operating at a MBT facility

Open systems may also rely on mechanical turning for aeration using machinery equipped with augers, paddles or tines (Plate 2.1) (EPA, 1995). Among the benefits of using turned windrows over static piles is the ability to add moisture if and when required. The disadvantage of using agitation only windrows is the difficulty in controlling the windrow temperature, with the temperature development typically displaying a saw-tooth pattern, rising steadily, and then dropping sharply after turning (Stentiford, 1996). Another disadvantage is the potential release of odour during the turning stage.

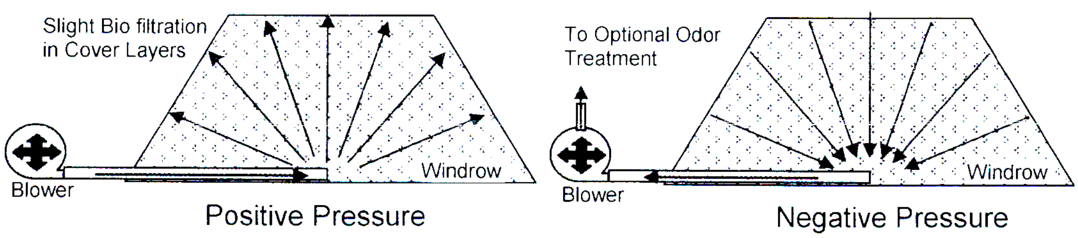

Figure 2.4 Forced Aeration is achieved either through positive or negative pressure

Airflow systems create movement of atmospheric air through the material, providing oxygen and removing compost gas. The airflow is maintained by forced aeration, or by exploiting the naturally occurring thermal convection and diffusion mechanisms (Mollekopf et al, 2002). Forced aeration - also termed active aeration because of the energy input requirements - uses blowers to drive the air flow. The blowers are connected to a system of pipes that run through the piles and provide either positive air-pressure to force air through the pile and out the sides, or negative pressure which draws air in through the sides of the pile. Hybrid systems that make use of both methods may also be used (Stentiford, 1996). The use of positive pressure prevents the treatment of emission gas, as is possible when using negative pressure systems. However, the outer layers of the pile may act as a bio filter and consequently produce fewer odours than negative pressure systems where the exhaust gas is untreated (Bidlingmaier, 1996) see Figure 2.3. The use of positive pressure also serves to distribute the heat from the inner regions to the outer edges of the pile (Williams, 1998).

The natural aeration systems rely on thermal convection and diffusion (Figure 2.4). The diffusion of oxygen into a pile of organic material is not an efficient means of aeration (Haug, 1993), and the rate of oxygen consumption often exceeds the rate at which the oxygen is replenished leading to the formation of anaerobic zones. The use of diffusion is restricted to piles of low height containing compounds with a low oxygen demand, such as brush (Mollekopf et al, 2002). Thermal convection occurs as a result of the buoyancy pressure created by the density differences between the hot compost gas and the cooler ambient air as the thermal energy generated by the aerobic degradation heats the gases within the compost mass. Based the Ideal Gas equation, the density of a gas is found to be inversely proportional to the temperature of that gas (Brown et al 1997). Thus the hot compost air is less dense than the cooler ambient air, resulting in the upward displacement of the compost gas. The driving pressure can be calculated according to the formula (2.3) (Mollekopf et al, 2002)

$$\Delta p = \Delta \rho * g * h \qquad 2.3$$

where:

Δp = driving pressure

$\Delta \rho$ = density difference between compost gas and ambient air

g = acceleration due to gravity

h = height over which density difference is effective

2.8.3 Dome Aeration Technology

The Dome Aeration Technology (Mollekopf et al, 2002) is a passive aeration system, utilising thermal convection to drive the aeration process (Figure 2.6). The principle of the technology is the use of structures to maintain large voids within the windrow. One structure, called a dome, is positioned in the centre of the windrow for the accumulation of hot compost gas, which is then vented to the atmosphere via a chimney. The other structure, called a channel, allows for the movement of ambient air into the windrow. The height of the column of hot gas is equivalent to the height of the dome and chimney, and thus a greater effective driving head is created. Another advantage is the use of the chimney for gas venting, and the channels for the induction of ambient air, negating any concerns of low permeability in the cover material (Mollekopf et. al., 2002). The domes and channels are constructed from construction steel bars and mesh, and the chimneys are generic PVC sewer pipes. The plant requirement for the windrow construction is flexible and equipment commonly found on landfills can be used (Mollekopf et. al., 2002).

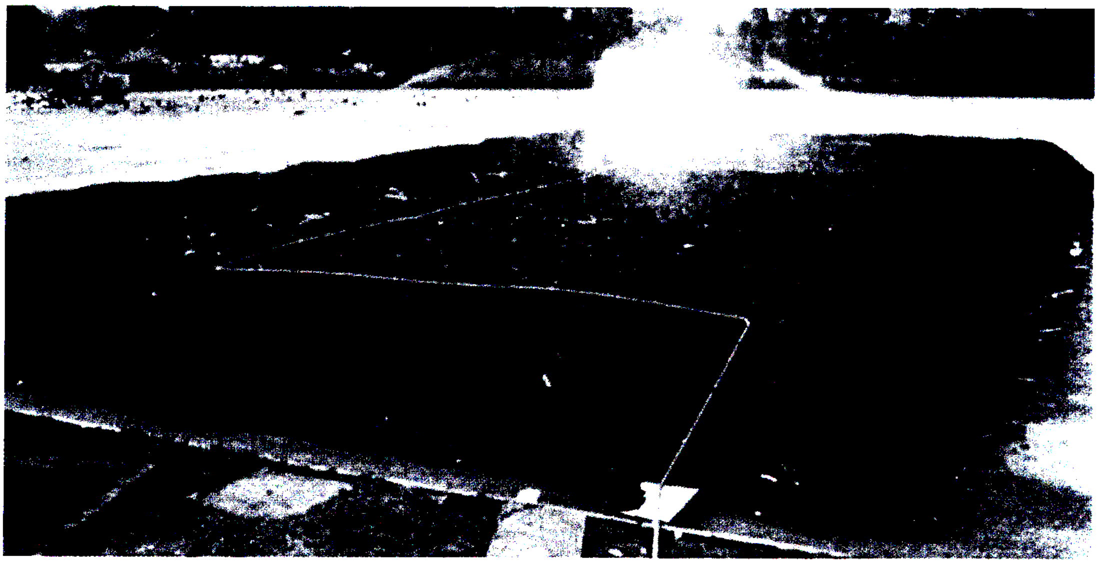

Plate 2.4 Trial Windrow In Brazil using sole aeration. Note sprinkler irrigation for moisture addition (Munnich et. al, 2006)

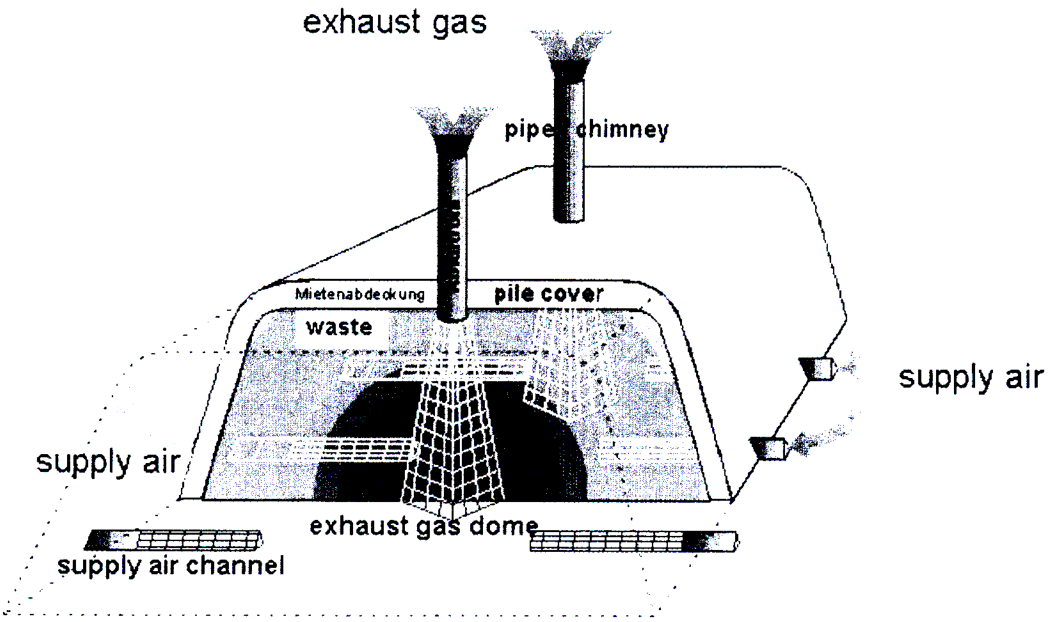

Figure 2.6 The Dome Aeration Windrow Technology (Brummack, 2004 in Kuehle-Weidemeier, 2004)

2.9 ENVIRONMENTAL PROBLEMS ARISING FROM COMPOSTING

A composting operation may present environmental problems that must be taken into account and the appropriate precautions should be taken. The major problems arise as a result of leachate, odour and pathenogenic microorganisms

Leachate

Free liquid that has been in contact with the rotting materials, and is released during the degradation process can pose problems such as groundwater contamination, the generation of odours, breeding of flies and mosquitoes (EPA 1995). Open windrow systems, will generate leachate when exposed to rainfall, and provision for leachate collection must be made.

Odour

Offensive odours and ammonia gas may be released during the initial intensive degradation stage. The odour intensity may increase if the operation conditions are not properly controlled. Means of controlling odour include routing exhaust air through filters, deodorizers or scrubbers. Bio filters are also an option for odour removal, and involve passing the odorous gas though a porous material such as finished compost, soil or sand, which acts as a substrate for microorganisms. Odour removal occurs as a result of adsorption/absorption by the material and as a result of oxidation by the microbes.

Pathogens

The spores of *Aspergillus fumigatus*, a fungus that occurs naturally in decaying organic matter, can cause health problems for some workers, particularly if the conditions are dry and dusty (EPA, 1995).

2.10 THE EFFECT OF MBT ON LANDFILLING OF WASTE

The major benefits of MBT are a reduction in waste volumes, biogas production and leachate toxicity. However, the degree of benefit is dependant on the following site-specific factors (Bone et. al., 2003):

- The extent of source separation
- The waste input
- The type of mechanical treatment
- The type and duration of the biological treatment

These factors will influence the nature of the material that is finally landfilled, and consequently influence the emission potential of the landfill. Results from lab scale and field sampling are presented for discussion.

Lab Scale Studies

Tests on the effect of MBT on waste emissions have been conducted by numerous authors, three of which are presented here. The results of two investigations, in which treated was subjected to a landfill simulation with biogas and leachate analysis, are presented in Table 2.6

Table 2.6 Results of Lab-Scale Landfill Simulation on MBT waste

Source			Muntoni et al., (2005)		Leikam et al. (1997)	
Composting Period	Unit	**Untreated**	**8 weeks**	**15 weeks**	**Untreated**	**16 week**
Lowest pH		5.5	7	7.1	5.5	6.8
Highest pH		7.5	8	7.7	7.7	7.5
COD Load	mg/kg	80000	25000	20000	27500	2000
NH4-N	mg/kg	570	250	150	>2000	200
Biogas Generation	l/kg	140	40	20	200	20

Note: Approximate values read off charts

The effect of MBT is clear in the absence of an extended acidic inhibition which has is characteristic of landfills containing untreated waste. The removal of COD and ammoniacal nitrogen is significant, with 75-90% reductions for 15-16 weeks of biological treatment. Biogas generation is also significantly affected with 85-90% reduction in biogas generation after 15-16 weeks. A shorter treatment period of 8 weeks results in a 70% reduction in COD, a 56% reduction in ammoniacal nitrogen and a 70% reduction in biogas.

Field Studies

A study by Robinson et.al., (2005) consisted of a review of leachates from literature and field sampling and provides a good summary leachate parameters recorded from full scale MBT and Municipal Sorted Organic Residue (MSOR) operations . The results of this work are presented in Table 2.7. From the results in Table 2.7 and Figures 2.6 and 2.7, the following observations can be made:

- Degree of degradation is dependant on the composting time,
- but the efficiency of degradation within this time is dependant on the process
- COD and ammoniacal nitrogen values are greatly reduced,
- but are still too high for discharge into the environment with results from some operations resembling that of mature leachate from untreated waste landfills (Robinson et al., 2005) according to current EU regulations

The high latent COD and reduction in ammoniacal nitrogen in leachates from treated wastes has been observed in many investigations and it is thought that the formation of humic substances is responsible for these characteristics (Ziegler, 1997; Muntoni, 2005).

Humic COD in MBT leachates

The conversion of COD to humic COD during the composting process has been demonstrated by Muntoni (2005) (Figure 2.8). When general treated waste is subjected to anaerobic conditions in an experiment, Muntoni (2005) reports that the humic COD remains a significant proportion (40%) of the COD in leachates. The stable nature of these compounds may be a disadvantage if treatment of this leachate is required (Robinson et al, 2005) but may also be advantageous if the humic compounds are shown to have a low environmental risk although further research into the toxicity of these leachates is required (Muntoni et al., 2005). In Great Britain, after lengthy negotiations with the Enivronmental Agency, consent was granted use a value of 30mg/l for the 5 day Biological Oxygen Demand (BOD_5) as an effluent limit and plants such as the Arply Leachate treatment works is fully compliant in terms of discharge standards, despite COD values in excess of 1000mg/l (Robinson et. al, 2008).

Table 2.7 Results of investigation into Leachate from MBT waste by Robinson et al., (2005)

Waste Input	MSOR*		MBT Waste Leachate – Robinson et. al. study (2005)						MBT Waste Leachate - Various Sources				
	Young	Mature											
Composting Process			Passive Windrow	Passive Windrow	Turned Windrow	Turned Windrow	Container	Container & Windrow	Not Specified				
Intensive	n.a.		0	0	0	0	16	2	4	4	2	16	3
Secondary			12	30	25	8	0	30	9	43	1	8	19
pH	6	8	8.1	8.3	7.9	7.9	8.4	8.5					
COD	150000	10000	582	4670	228	1620	869	1020	2780	1170	540	4000	1900
BOD5	100000	4000	46	202	3	35	6	3	52	9	158	111	14
TOC	50000	4000	180	1480	78	543	308	340					
NH_4-N	4000	4000	195	1130	286	197	34.2	1.8	197	11	56	292	340
NOx	<1	<1	10.3	<1	16.1	<1	7.3	5.0					

* Source-Term for MSOR Leachate given by Robinson et al., (2005)

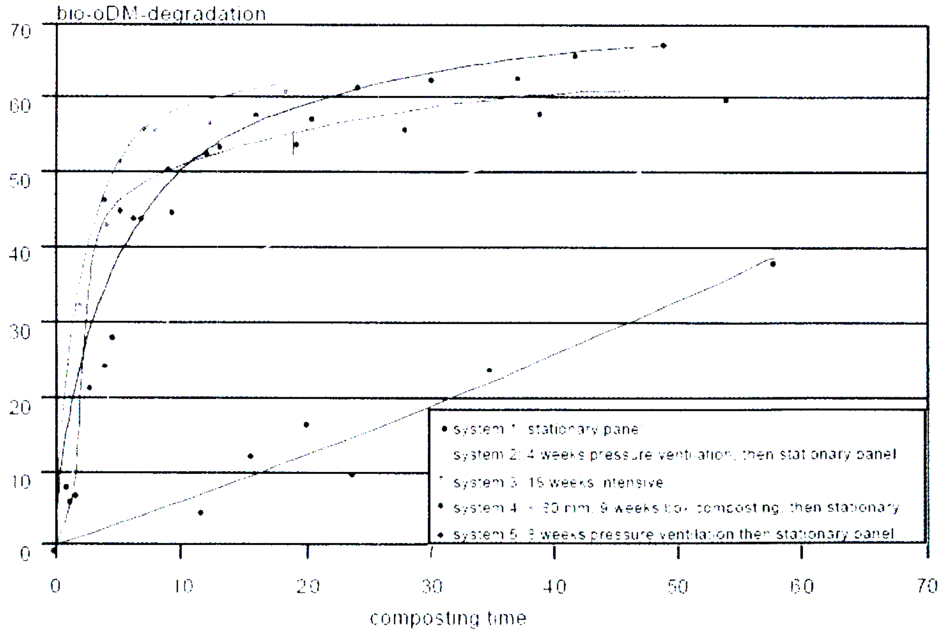

Figure 2.7 Biological degradation vs composting time for various composting processes (Fricke et. al (1999) in Muller et. al. (2002))

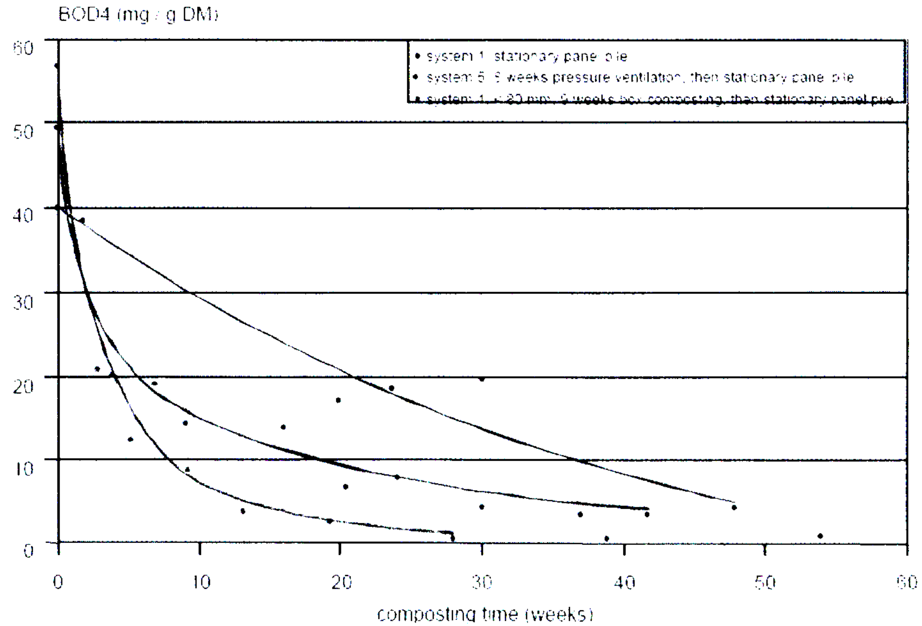

Figure 2.8 BOD4 vs composting time for various composting processes (Fricke et. al (1999) in Muller et. al. (2002))

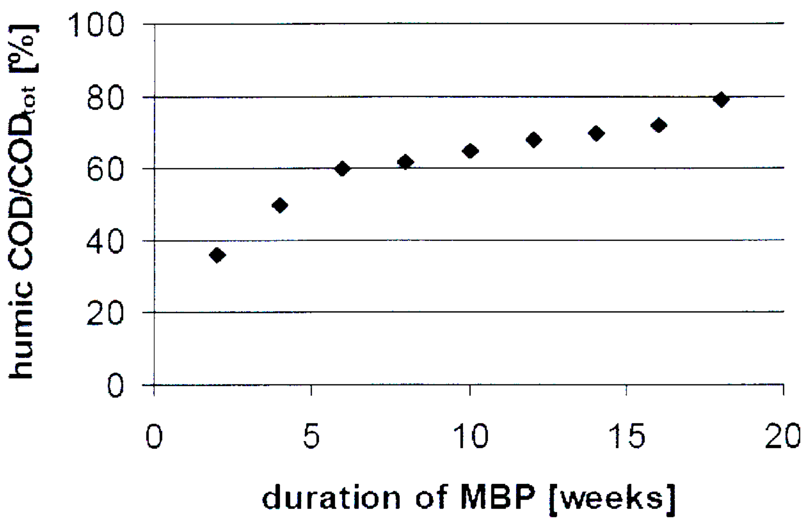

Figure 2.9 Change in the Proportion of Humic COD to Total COD over composting time (Muntoni, 2005).

Humic Nitrogen in MBT leachates

The question has been raised of the fate of organic nitrogen (N_{org}) during composting processes. Studies have shown significant reductions in Total Kjeldhal Nitrogen (TKN) in leachates from MBT wastes, however the nitrogen oxide (NO_x) levels remain low (Robinson et al., 2005). Muntoni (2005) reports that the conversion to ammonia gas during the composting process does not account for the reductions in TKN values. However, by studying the levels of humic nitrogen (N_{humic})during composting, the conversion of organic nitrogen to humic forms becomes evident (Figure 2.10). The long term fate of this sequestered humic nitrogen in the anaerobic landfill environment remains unknown (Robinson et al., 2005).

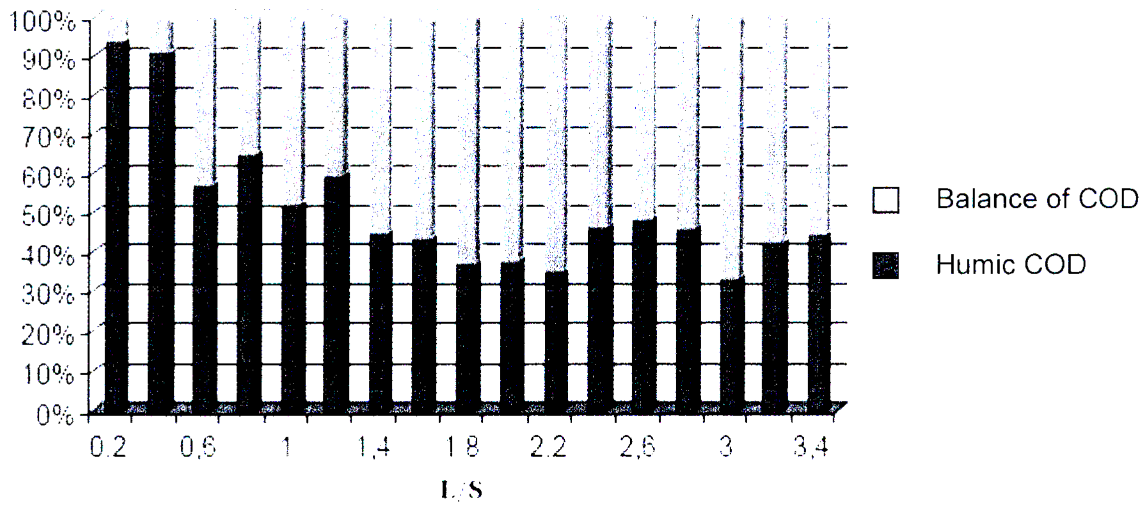

Figure 2.10 Change in the Proportion of Humic COD to Total COD in anaerobic conditions (Muntoni, 2005)

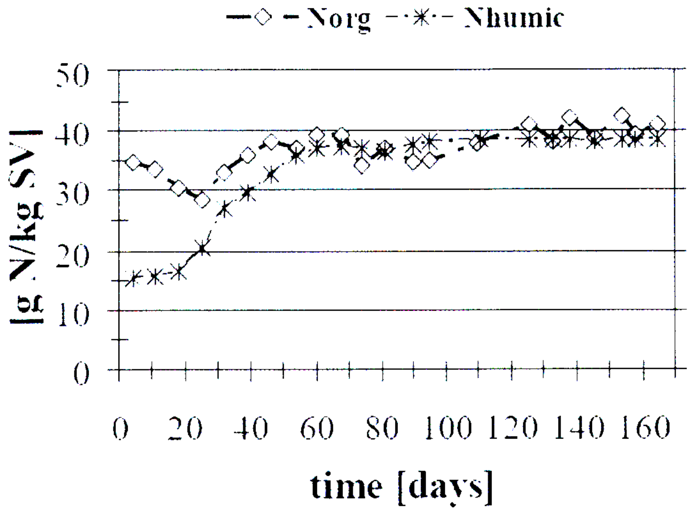

Figure 2.11 Change in the proportion of humic Nitrogen with composting time (Muntoni, 2005)

Physical Characteristics of MBT Output

Application of the MBT process results in a significant change in the physical characteristics of municipal solid waste. Removal of recyclable materials from the waste stream by sorting, reduction of waste volumes by shredding, reduction of the organic fraction and moisture content of the waste by biological treatment can reduce the waste volume by 20-40% (Leikam et al., 1997; Muntoni, 2005). The majority of the output is less than 80-100mm (Muntoni, 2005), although the volume of the material greater than 60mm is significantly greater than the mass suggests due to the lower density of these larger fractions (Kuehle-Weidemeier, 2003). The action of removing all the large fraction materials may have the undesirable consequence of reducing the efficiency of the biological treatment step by reducing the void volume and moisture retention of the waste and emphasis must still be placed on optimisation of the biological treatment step (Brummack, 2005). The calorific value of the waste, an important consideration if the process is used in conjunction with thermal treatment, varies according to the particle size, as shown in figure 2.11. The compactability of the waste is also improved and landfilled waste densities can increase from 0.8-0.9 tons/m^3 to 1.2-1.4 tons/m^3 (Leikam et al., 1997; Scheelhaase et al., 1997; Kuehle-Weidemeier et al., 2003). The total saving of landfill airspace may therefore be as much as 50-60% (Heerenklage et al., 1995; Muntoni, 2005) and is significant when considering the social complications and construction costs in constructing a new landfill site. The current cost of landfill lining systems ranges from R250-R350/m^2, depending on the availability of materials. Considering that the depth of landfilling in the Durban area ranges from 10-30m, the savings in terms of landfill airspace utilisation are R10-R35/m^3 (Pass, 2008).

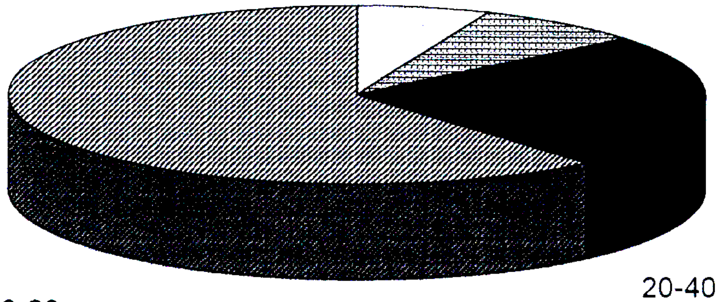

Figure 2.12 Particle Size of MBT Output (Kuehle-Weidemeier, 2003)

This enhanced compaction results in a reduction in waste permeability, with hydraulic and gas permeabilities of 10^{-9}m/s and 10^{-17}m^2 respectively (Muntoni, 2005; Bauer, 2006). The altered nature of the treated waste has implications on the choice of landfill plant with sheeps-foot and vibratory rollers achieving greater compaction rates than standard landfill compactors. An operational MBT waste landfill is shown in plate 2.3.

Table 2.8 Changes in Physical Parameters of Waste After Waste Treatment

Parameter	**Unit**	**Untreated**	**MBT**
Physical Characteristics			
Mass Reduction	%	na	20-40
Placement Density	t/m^3 (wet)	0.8 - 0.9	1.2-1.4
Hydraulic Conductivity	m/s	$10^{-3} - 10^{-6}$	10^{-6} 10^{-9}
Gas Permeability	m^2		10^{-17}
Airspace Reduction	%	na	50-60

Biogas Control at MBT Landfill sites

Studies indicate that the low generation of biogas coupled with the low permeability of landfill sites containing treated waste does not warrant the installation of gas extraction equipment (Stegmann et al., 2001, Muntoni et al., 2005). The oxidation of methane in the landfill cover layers (Figure 2.12) has been shown to be an effective means of gas management and odour control (De Visscher et al., 2003; Muntoni et al., 2005) These specialised layers should consist of porous material of a depth suited to the expected rate of biogas generation with a 30cm layer of soil able to oxidise 0.003m^3 CH_4/m^2/h. A MBT landfill cap proposed by Stegmann et. al, (2001) which does not include any impermeable barriers is shown in Figure 2.13.

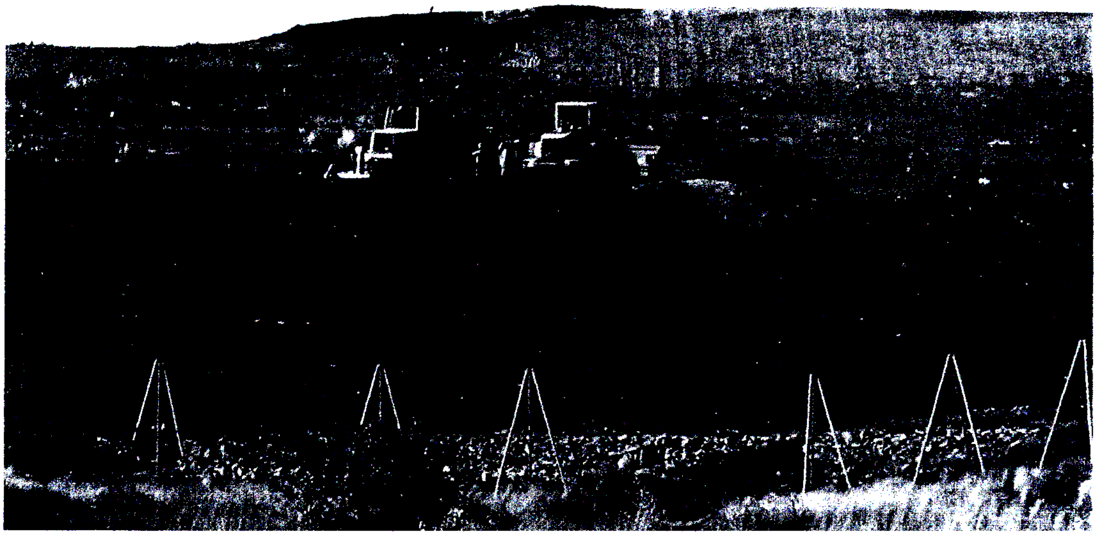

Plate 2.5 An operational MBT waste Landfill in Germany

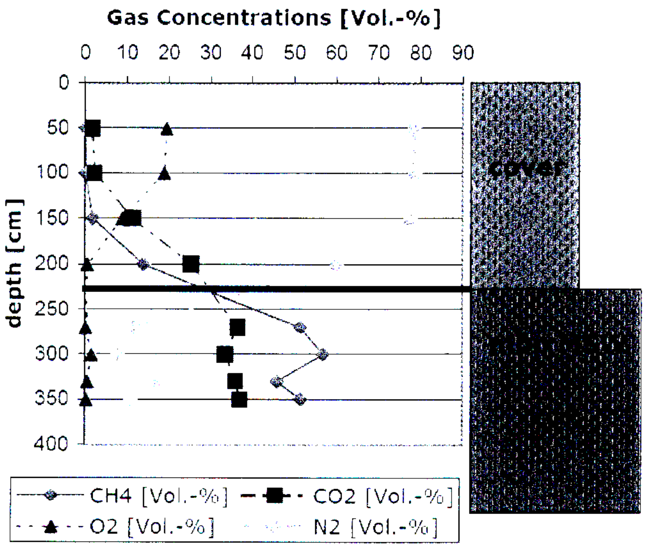

Figure 2.13 The oxidation of Methane in a Soil Cover (Felske et al., 2003)

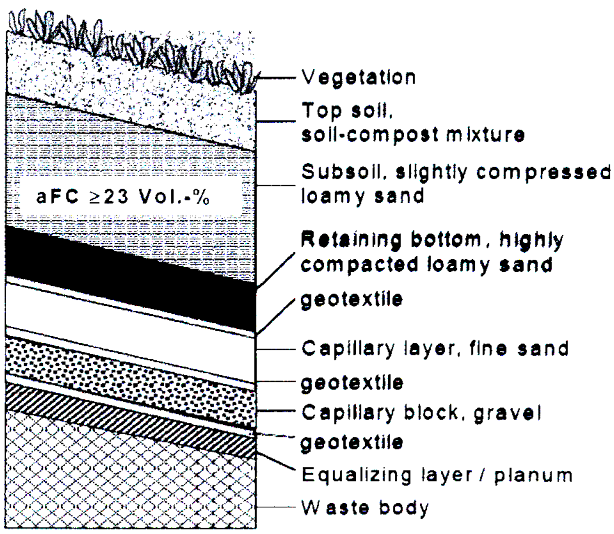

Figure 2.14 MBT Landfill Cap Proposed by Stegmann et al., (2001)

2.11 THE DEVELOPMENT OF APPROPRIATE SOUTH AFRICAN SOLUTIONS

The current activity of landfilling untreated waste poses many environmental problems and is an unsustainable practice. Currently the South African legislation is focused on the restriction of leachate formation, and allows for the landfilling of untreated waste.

However the current environmental legislation may change. To quote the then minister of Water Affairs and Forestry, Mr Kader Asmal,

"It is essential that at all levels we should consider how to address the waste management problem, in order to conform to international standards and to protect the constitutional right of every South African to a clean environment"
(Asmal, 1996 in Strachan, 1999).

The Department of Water Affairs and Forestry (DWAF) have therefore indicated that international standards are what South Africa must strive to achieve. However, a further challenge that is faced is achieving these standards with greater resource limitations. An appropriate solution is thus needed, that improves the current conditions in an affordable manner, and which benefits both the environment and the human population.

The Polokwane Declaration

Strives have been made in order to reduce the negative impact that landfilling of untreated waste poses. At the first National Waste Summit, held in Polokwane in 2001 with representatives from government, business, industry and civil society, the urgent need to protect the environment through pollution prevention was recognised. The focus was on the reduction, reuse and recycling of waste, promoting the hierarchal approach to waste management. The following goal was agreed upon:

To reduce waste generation and disposal by 50% and 25% respectively by 2012, and to develop a plan for zero waste by 2022.
(Wiechers et. al., 2002).

Despite the inconsistency in this statement; i.e. a 50% reduction in waste generated should realise at least a 50% reduction in waste disposal, the ultimate objective of waste reduction is clear. This principle of waste reduction aligns well with the implementation of MBT, where the mechanical processing stage allows for the implementation of materials recovery, and where the aerobic stabilisation of organic waste allows for potential reuse as compost.

The Mechanical Biological Treatment of waste is an option that would alleviate the environmental problems posed by landfill and represents a way forward in terms of achieving the goals set by the Polokwane declaration. However, not all these technologies are appropriate for use in South Africa, as high capital and running costs render the technology unviable, particularly with the prospect of application in rural areas and an appropriate technology is required. Factors such as the low cost of the structures, zero energy input during the composting period, potential high-labour based operations and the fact that standard landfilling equipment can be used for the operations, eliminating the need for a large capital investment, are beneficial for the establishment of a pilot scale MBT operation. It is in this context that the DAT was found appropriate and was therefore selected for this investigation.

2.12 MBT COSTS

The cost of MBT operations varies according to the technology employed and the country in which the project is implemented. The cost of MBT required for EU standards is some 40-60 €/ton while low cost technology costs approximately 10-20€/ton (Kuehle-Weidemeier 2005). This cost excludes the cost of landfilling the residues. For a study in Brazil, a developing country, Munnich et al., (2006) compiled a comparison of MBT costs for low and high tech plants in developing countries and Germany (Table 2.9)

Table 2.9 MBT capital and operation costs in developing Countries and Germany (Munnich et al., 2006)

MBT Technology	**Low Tech Plant**		**High Tech Plant**	
	Investment Costs	Operation and Maintenance	Investment Costs	Operation and Maintenance
Developing Countries	\$10~30/t	\$8~12/t	\$80~220/t	\$15~30/t
Germany	\$40~100/t	R150~250/t	\$250~450/t	>\$50/t

Considering the current cost of landfilling in the Durban region of approximately R200/ton (approximately \$20 per ton) (Strachan, 2008), and the lack of regulations controlling the output of MBT plants, the low technology option may be feasible locally. High tech plants may be beneficial in terms of more efficient operations and lower emissions; however the cost of waste disposal will increase substantially. If costs are a limiting factor in the establishment of a MBT plant, the Clean Development Mechanism may be a source of funding derived through the sale of Certified Emission Reductions on the global carbon market.

2.13 THE CLEAN DEVELOPMENT MECHANISM

The realisation that man's activities which result in the release of greenhouse gas is causing an unprecedented change in our earth's climate has led to the establishment of the United Nations Framework Convention on Climate Change (UNFCCC) and the Kyoto Protocol, signed in 1997, as a step towards the reduction of greenhouse gases released to the atmosphere. The requirement is for developed countries (Annex 1) to reduce their emissions to agreed levels while developing countries (non Annex 1) are only required to monitor emissions. South Africa acceded to the Kyoto Protcol in 2002 and was categories as a Non-Annex 1 (or developing) country under the terms of the Protocol (http://www.environment.gov.za/ClimateChange2005/Kyoto_Protocol.htm) which allows for Clean Development Mechanism projects in this country.

The Clean Development Mechanism, or CDM, is the only Kyoto Mechanism (Article 12) that addresses developing countries. CDM aims to "*direct private sector investment in emission reductions projects in developing countries*" and may be a valuable means to achieve the implementation of projects where financial viability a critical drawback.

The two primary objectives of the CDM are:

- To assist Annex I countries in reaching their emission reduction targets, and;
- To promote sustainable development objectives in the host countries (non-Annex I countries).

CDM projects must meet the following requirements:

- the participation in a CDM project activity is voluntary (not regulation)
- the project must incur measurable reductions in emissions
- the reduction in emissions must be additional to any that would occur in the absence of the approved project activity
- the project must contribute to sustainable development in the host country

The eligibility of a MBT project is addressed according to each of these requirements:

The activity is Voluntary

This is true for the case of MBT where the treatment of general waste is not a legislative requirement according to the current regulations which have been promulgated in the minimum requirements (DWAF, 2003).

Measureable Reductions in Emissions

The UNFCCC has assigned a global warming factor of 21 to methane. Thus the avoidance of methane generation through the aerobic stabilisation of waste, as occurs during biological treatment, is a measureable reduction.

The Emission Reductions are additional

The absence of local legislation requiring the extraction and destruction of landfill gas methane or the treatment of solid waste and the fact that the implementation if such projects face a great financial hurdle means that the project should be additional.

The project contributes to sustainable development

The Marrakech Accords (UNFCCC, 2002), which details the requirements for CDM projects states that the sustainable development criterion is the prerogative of the host country. The Department of Minerals and Energy, which is the South African Designated National Authority has established the following sustainable development criteria (http://www.dme.gov.za/dna/dna_susdev.stm):

- Economic: Does the project contribute to national economic development?
- Social: Does the project contribute to social development in South Africa?
- Environmental: Does the project conform to the National Environmental Management Act principles of sustainable development?

These criteria can all be fulfilled with proper project development that takes these requirements into account.

Evaluation of the eligibility of MBT in terms of emission reductions is made easy when recognising that internationally, biological waste treatment projects have been registered as CDMs under the approved baseline and monitoring methodology AM0025 - **"Avoided emissions from organic waste through alternative waste treatment processes"** using the Methodological tool **"Tool to determine methane emissions avoided from dumping waste at a solid waste disposal site"**

Thus in the South African context where MBT and gas management is not regulation and where it is possible to fulfil the DNA's sustainable development criteria, South African MBT projects have the potential to become Clean Development Mechanisms with potential revenues from the sale of CERs a way to improve the financial viability of the project.

2.14 RESEARCH INTO MBT IN SOUTH AFRICA

From the review of literature, there is a substantial pool of knowledge in the field of MBT in Europe where the technology has been implemented at full scale and is a legislative requirement for the disposal of solid waste. Investigations in developing countries have also been conducted with research conducted by (Munnich et. al, 2006) and others. However, there are few studies on MBT waste in South Africa. The University of Kwa-Zulu Natal (UKZN) and the Tshwane University of Technology and Technical University of Dresden have conducted collaborative research on the subject since 2001 and the research into the DAT windrow system for aerobically treating waste, and the study of the emissions thereof was the first of its kind in South Africa. Journal Publications of this work is available in Appendix E.

The DAT windrow system and the operation at the Cottbus Landfill site are described in the following chapter.

CHAPTER 3

DOME AERATION TECHNOLOGY

The Cottbus Landfill Site – A Case Study

3.1 INTRODUCTION

Research into the DAT system of waste treatment, through passive aeration in windrows, was initially conducted at the landfill site of Plauen in Germany. The research initiated by the University of Dresden (Paar, 2000) proved that the DAT is an effective means of windrow aeration. The system has been successfully implemented at full scale at the Cottbus Landfill Site which receives approximately 50,000 tons of refuse annually (Mollekopf, et. al. 2002) (Plate 3.1). This technology was brought to South Africa for the aerobic treatment trial conducted in this study and presented in Chapters 4 and 5.

The DAT system involves the creation of large airspaces within the windrow using steel mesh structures, called domes and channels. From the perspective of the windrow cross section, the domes are placed vertically in the centre of the pile, extending throughout the windrow height, and two channels are placed horizontally at the base, on opposite sides of the pile, extending from the edge towards (but not reaching) the centre. The domes and channels are placed in a staggered configuration relative to each other. A chimney extends from the top of the domes, and a layer of inert matter covers the composting material (Figure 3.1).

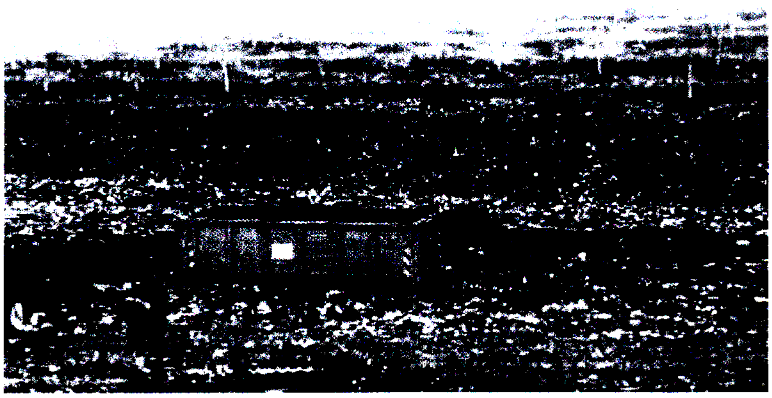

Plate 3.1 Rows of DAT Aerobic Pretreatment Windrows at the Cottbus Landfill Site (Source www.landfill.co.za)

The windrows used at Cottbus were 11m wide and 3m high, including 0,5m of cover material (Paar, 2000). Although the length of the windrow is not specified, generally 8 domes are used (approximately 40m long). The windrow cross-section and plan are shown in Figure 3.1 below.

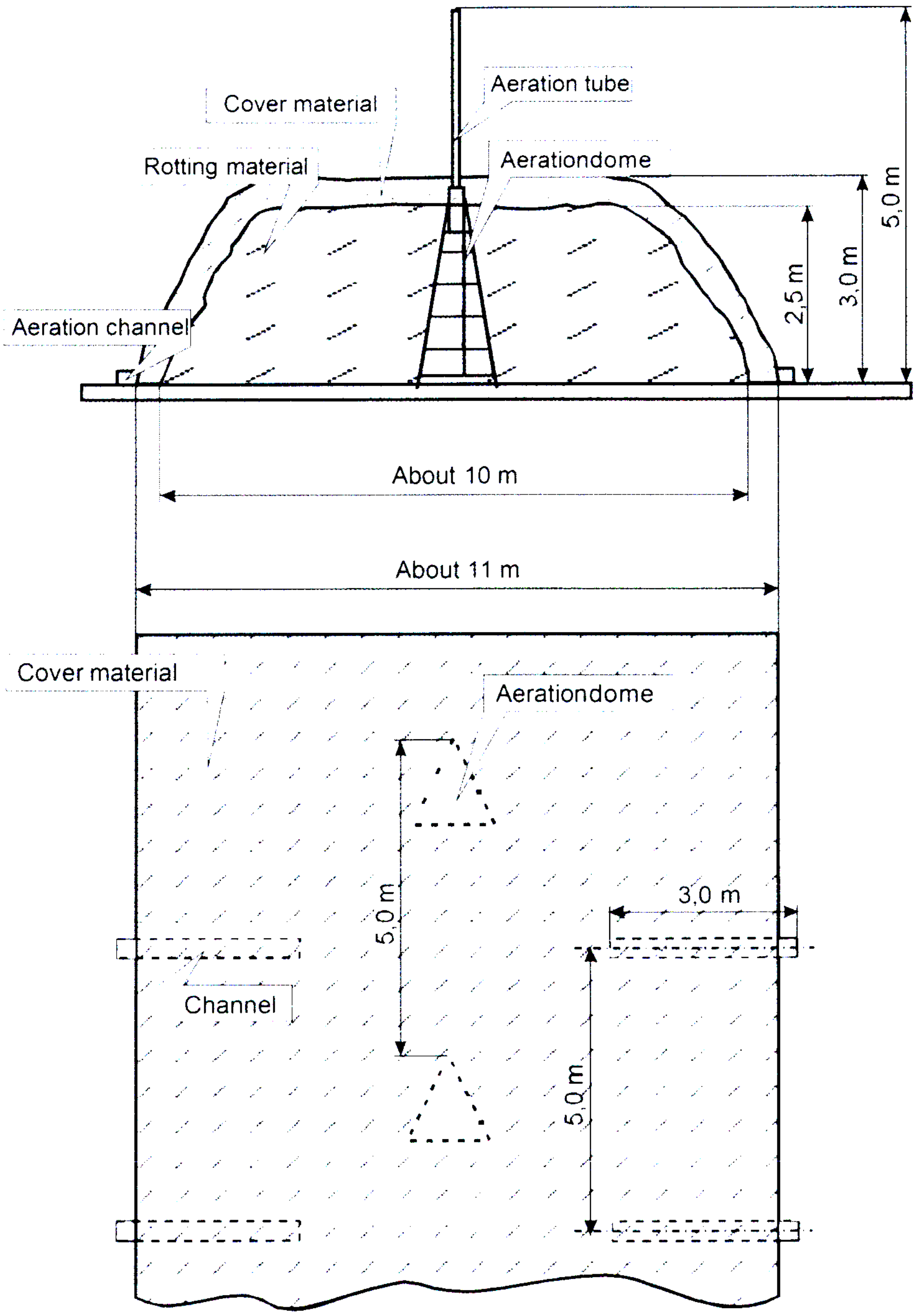

Figure 3.1 Typical DAT Windrow Layout (Paar, 2000)

3.2 OPERATION OF DAT MBT PLANT AT COTTBUS LANDFILL SITE

The site operations entail a three step mechanical pretreatment stage of mixing, shredding and wetting, followed by composting in DAT windows for four months (Paar et al, 1999, Mollekopf et. al., 2002). After the composting is complete the material is screened. Some of the material is reused for cover or optional coarse fraction recycling, and the rest is landfilled. The process is outlined in Figure 3.2.

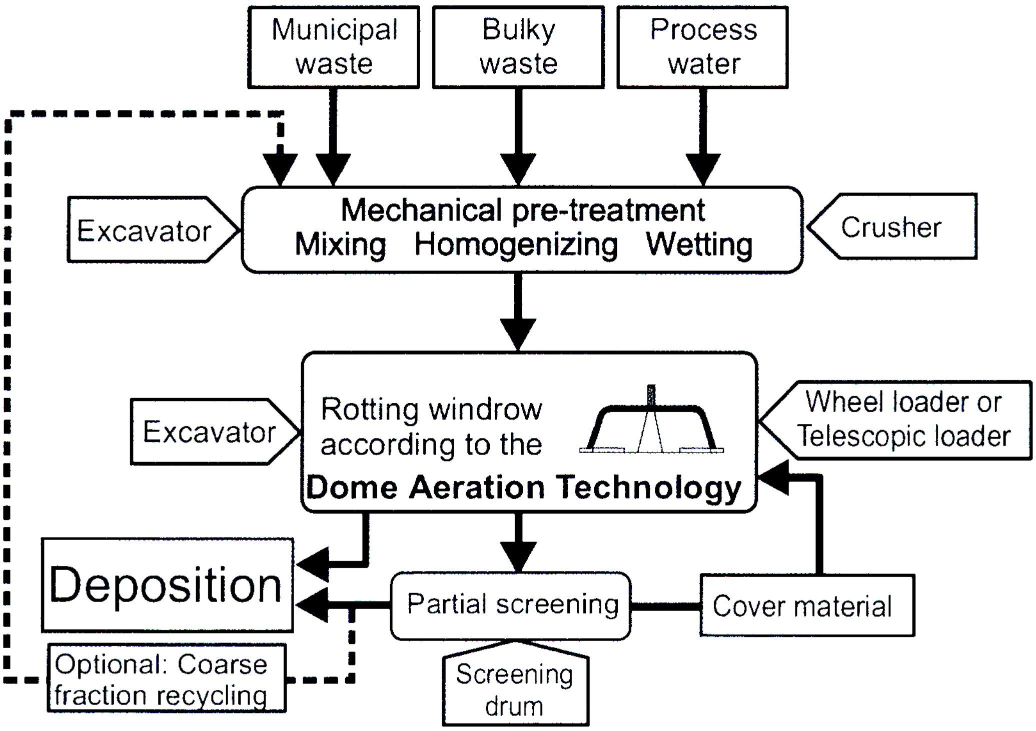

Figure 3.2 The MBT operations utilizing DAT windrows (Mollekopf et al, 2002)

3.2.1 Process Outline

According to Brummack et al. (2005) the effective operation of the DAT windrows requires adequate pore space in the waste pile. This is achieved by including material that provides structural support within the waste, termed structural material (SM) or bulky waste. The SM comprises materials that have a structural rigidity so to prevent the general waste from compressing onto itself, and thus maintaining the pore space. Although any material that fulfils this function is suitable, the SM is generally comprised of woody waste (branches etc). If the material is porous it also serves for the provision of moisture for the duration of the composting process. The MSW and SM are then shredded to a suitable particle size in order to increase the biodegradation surface, and also to ensure that sealed objects (eg plastic bags) are broken open. This reduces the occurrence of anaerobic zones within the windrow, where the generation of methane and offensive odours can take place. The general and bulky waste is loaded into a shredder simultaneously in a ratio of 2 parts MSW to one part SM thereby mixing the two material streams. While shredding, the moisture content of the waste is increased to approximately 55%. The adequate provision of moisture is crucial to the process as the windrows are left undisturbed throughout the composting period, and addition of moisture once the windrows have been constructed is not possible (Mollekopf, 2002). The windrows are then constructed on top of the landfill body, and for 4 months of composting the windrow is left undisturbed. Upon completion of the composting period the windrow is dismantled. A portion of the product is screened to recover fines for cover material for the next windrows (SM may also be recovered if needed). This cover material provides for heat insulation, reduces moisture losses and acts as a biofilter for the reduction of odours. The remaining composted waste is then landfilled (Paar et. al., 1999).

3.3 MECHANISMS OF DAT AIRFLOW

As the aerobic biological activity proceeds, the temperature of the windrow rises and the hot gases within the dome are driven up the chimney through buoyancy forces, creating a negative pressure within the windrow dome. Ambient air is drawn into the windrow via the side channels, and passes through the compost material and into the dome (Figure 3.3).

3.3.1 Driving Pressure

The pressure difference driving the air-flow in the windrow (Equation 2.3) can be derived by calculating the pressure difference between the windrow gas and the ambient air.

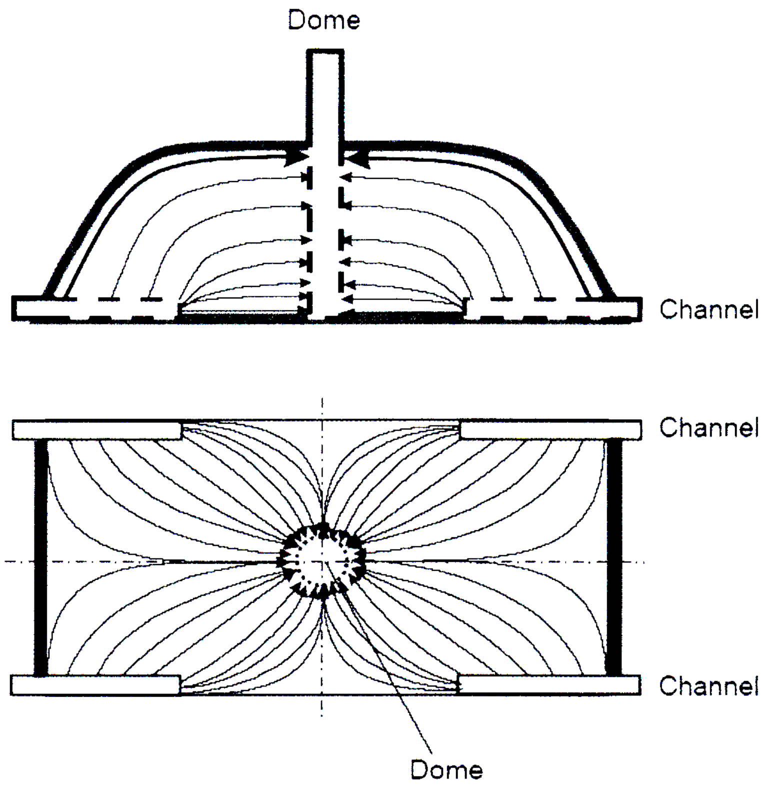

Figure 3.3 Stream-flows of DAT Aeration (Paar, 2000)

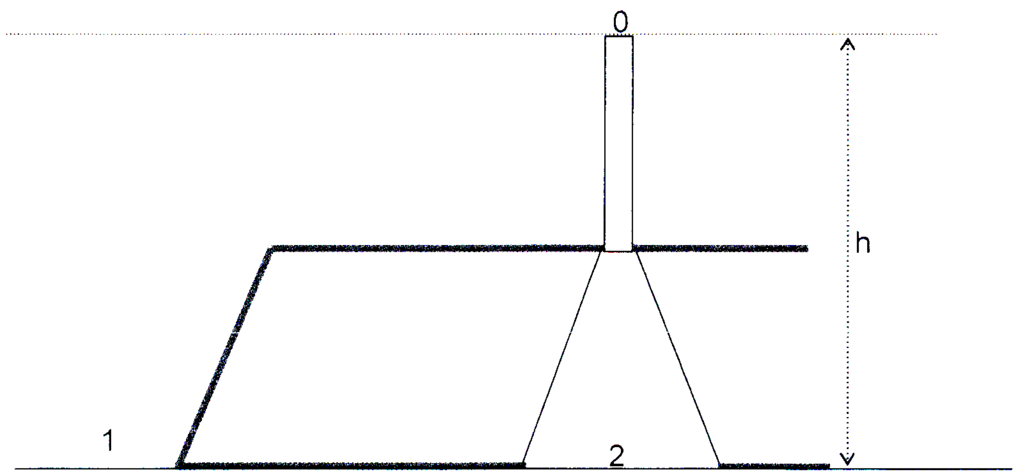

Figure 3.4 Boundary conditions for the determination of the pressure difference within the windrow.

domes, constructed from rebar and steel mesh, are three-sided pyramids, with a top section designed to support the chimney (Figure 3.7). The channels are triangular cylinders, constructed from rebar and steel mesh. The top of the channel is reinforced using steel plate, or structural steel sections, in order to support the weight of the overlying layers of windrow material (Figure 3.6).

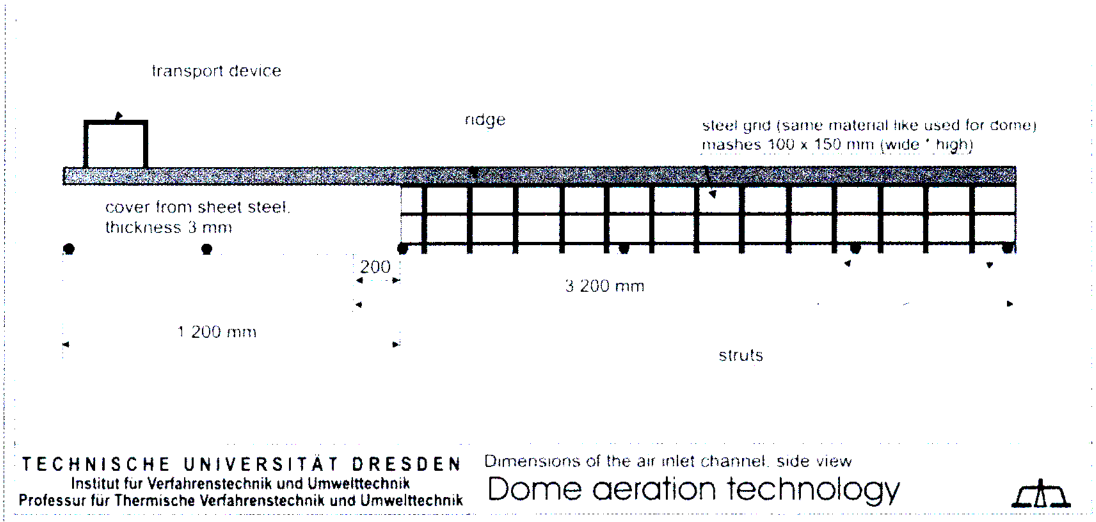

Figure 3.6 DAT Channel (Courtesy of Dr J Brummack)

3.5 THE PERFORMANCE OF THE DAT

The performance of the process can be evaluated by studying the windrow temperature, oxygen and carbon dioxide content of the exhaust gas, and the decrease in the TOC content of the material before and after the composting process. The results presented are based on the research conducted at Plauen (Paar, 2000).

Total Organic Carbon

The degree of waste stabilization is indicated by the change in the TOC content of the material from eluate tests before and after composting. The TOC in the eluate of untreated waste was measured to be 2500mg/l (Mollekopf et. al., 2002). After treatment for 4 months, the TOC content dropped to approximately 270mg/l (range 150-530mg/l) - a reduction of 90%. After six months of composting, the TOC dropped to 200mg/l (range 120-300mg/l). Thus the limiting value of 250mg/l for TOC, as stipulated by the German regulations is met within 6 months of composting (Mollekopf et. al., 2002).

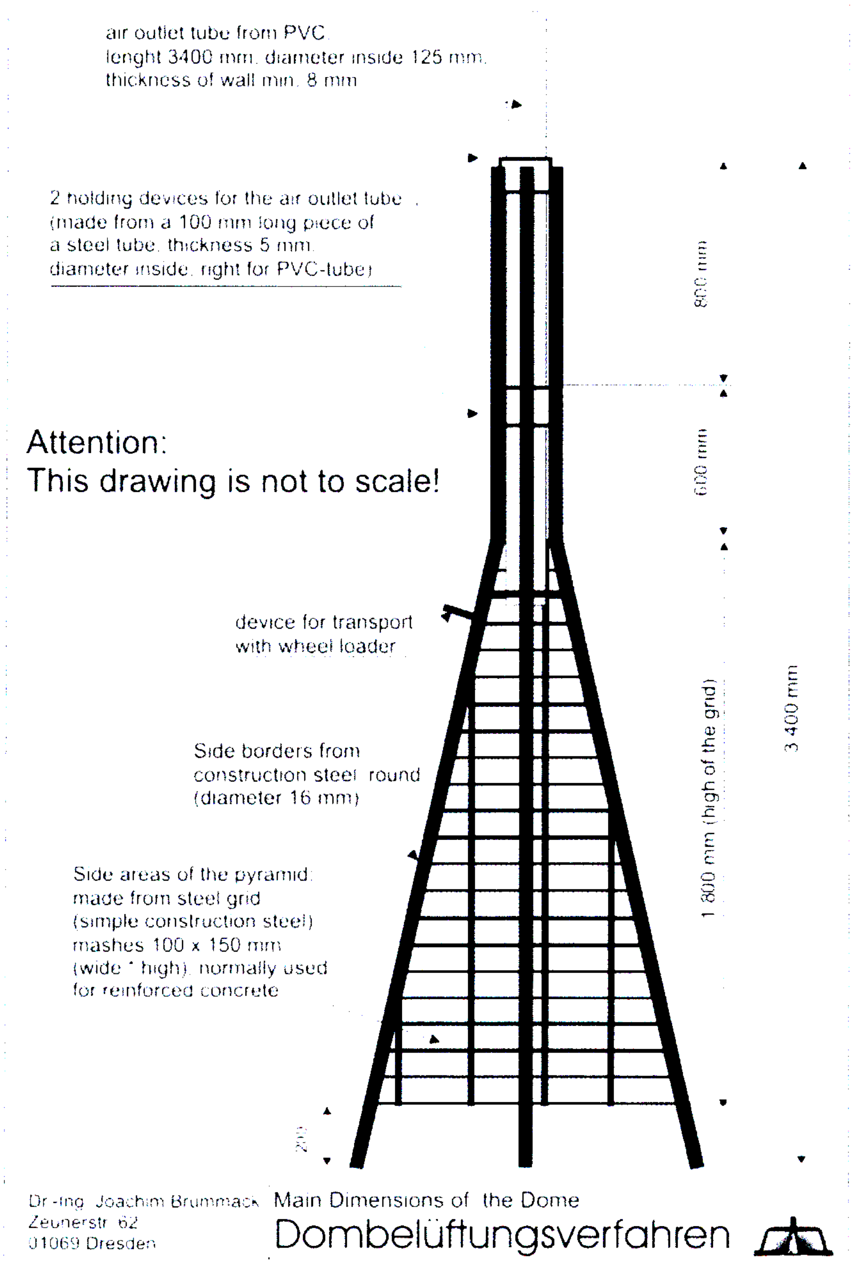

Figure 3.7 DAT Dome (Courtesy of Dr J Brummack)

Windrow Temperature

The Cottbus windrows reach the peak thermophillic activity after approximately 5 days, at temperatures of 70-75 °C. The temperature then continues to drop gradually over a period of 4 months to 35°C (Figure 3.8).

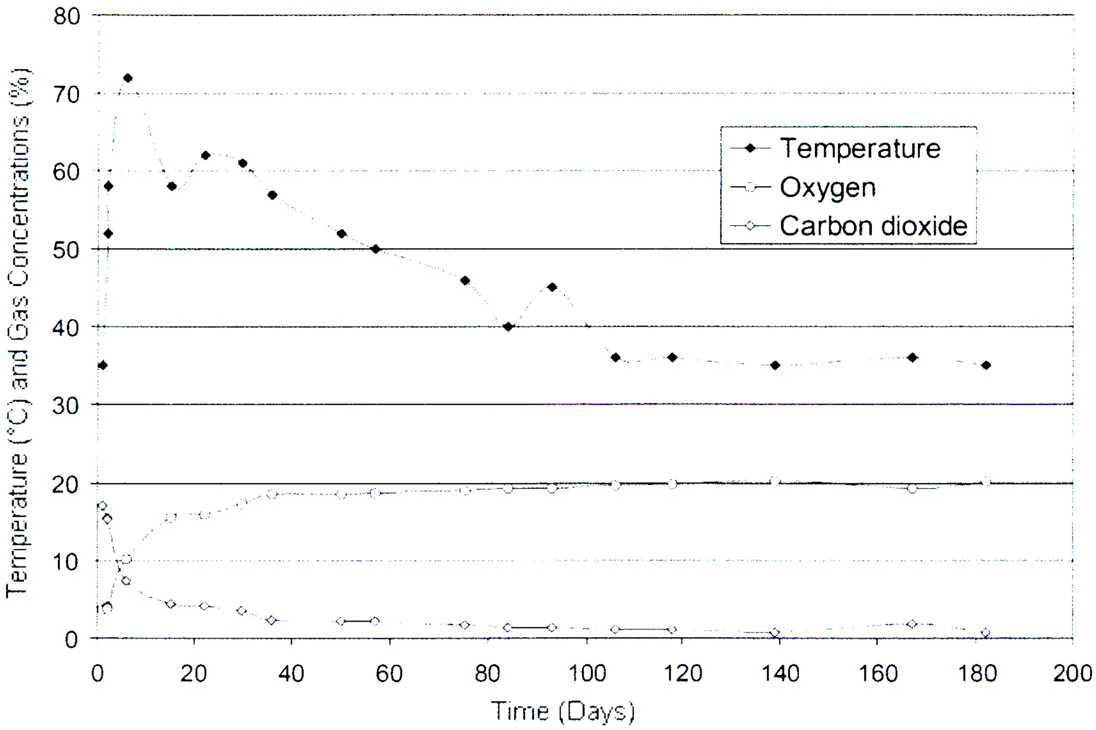

Figure 3.8 Temperature, Oxygen and Carbon Dioxide Concentrations from the dome of a DAT Windrow (Mollekopf et al, 2002)

Oxygen Content

Oxygen concentrations that lie within or are greater than 10-15% recommended for the maintenance of aerobic conditions. This is achieved within the first 5 days of composting, and remains above this value for the entire duration of the composting process (Figure 3.8).

Carbon Dioxide Production

The objective of the aerobic composting stage is to stabilize the biodegradable organic material through the activity of aerobic microorganisms which convert the organic compounds into carbon dioxide and water. The cumulative carbon dioxide production for the windrows studied by Paar (2000) amounted to 3000 to 3800m^3 per dome after 180 days of composting.

3.6 EQUIPMENT REQUIREMENTS AND COST OF THE DAT SYSTEM

The DAT operation makes use of the following equipment:

- Grab crane for loading material into the shredder
- Shredder equipped with sprinkler system for size reduction, mixing and wetting
- A truck for transporting and offloading windrow input
- A front end loader or wheel loader for windrow construction and
- An excavator or telescopic loader for final windrow shaping and post treatment dismantling
- A screen for fine material for windrow cover and optional post-treatment recover of coarse fractions
- A foreman to assist the machine operator in the placement of the domes.

According to Paar (2000), the combined operational and capex cost of the DAT process is 35-40DM. Considering a DM-€ exchange rate of 1.956 DM/€ and an inflation of 1.5%pa, the costs should be approximately 20-30€/t of material treated, assuming a 3-use cycle of the domes and channels but excluding the landfilling cost.

The DAT system has proven to be an effective means of mechanical biological waste treatment in Germany where it was developed. The low cost and availability of local materials and machinery required for the implementation makes the DAT system an attractive option for a developing country that wishes to participate in MBT projects. The implementation of the DAT for aerobic windrow treatment in Durban, South Africa, is presented in the following chapter.

CHAPTER 4

APPLICATION OF THE DOME AERATION TECHNOLOGY TO SOUTH AFRICA

4.1 INTRODUCTION

The Dome Aeration Technology for the aerobic pretreatment of waste was found to be an appropriate technology for Durban due to the relatively low construction costs, and the lack of energy and resource input required for the duration of the composting period. Problems that were expected to arise from the application of the technology were due to the differences between Germany and South Africa, in terms of climate, waste stream and equipment resources.

Three full scale trial windrows constructed at the Bisasar Road Landfill site in Durban and served to evaluate the performance of the process, highlight potential downfalls, and better understand the requirements for the application of aerobic treatment to Southern Africa. Material from the windrows was then tested in lysimeters to determine the behaviour of the treated waste in an anaerobic environment such as a landfill. The results from the windrow trials and anaerobic emission characterisation were then used to draw up a recommended waste disposal strategy. The study of aerobic waste treatment specific to the South African climate and the study of the emissions of the treated waste are original contributions towards the field of waste science in South Africa. The research process is outlined in Figure 4.1.

Three windrows were constructed and analysed between July 2002 and August 2003 and are named Windrow 1, 2 and 3, in order of construction. Windrow 1 was constructed in order to gain experience in the construction of the DAT windrows with moisture content measurements. Windrows 2 and 3 were constructed in order to provide a more detailed analysis of the gas and temperature evolution during the DAT composting process. Parameters such as windrow gas composition (oxygen, carbon dioxide and methane), temperature, and waste eluate characteristics were analysed to evaluate the windrow performance. The output material from these windrows was used for anaerobic lysimeter trials.

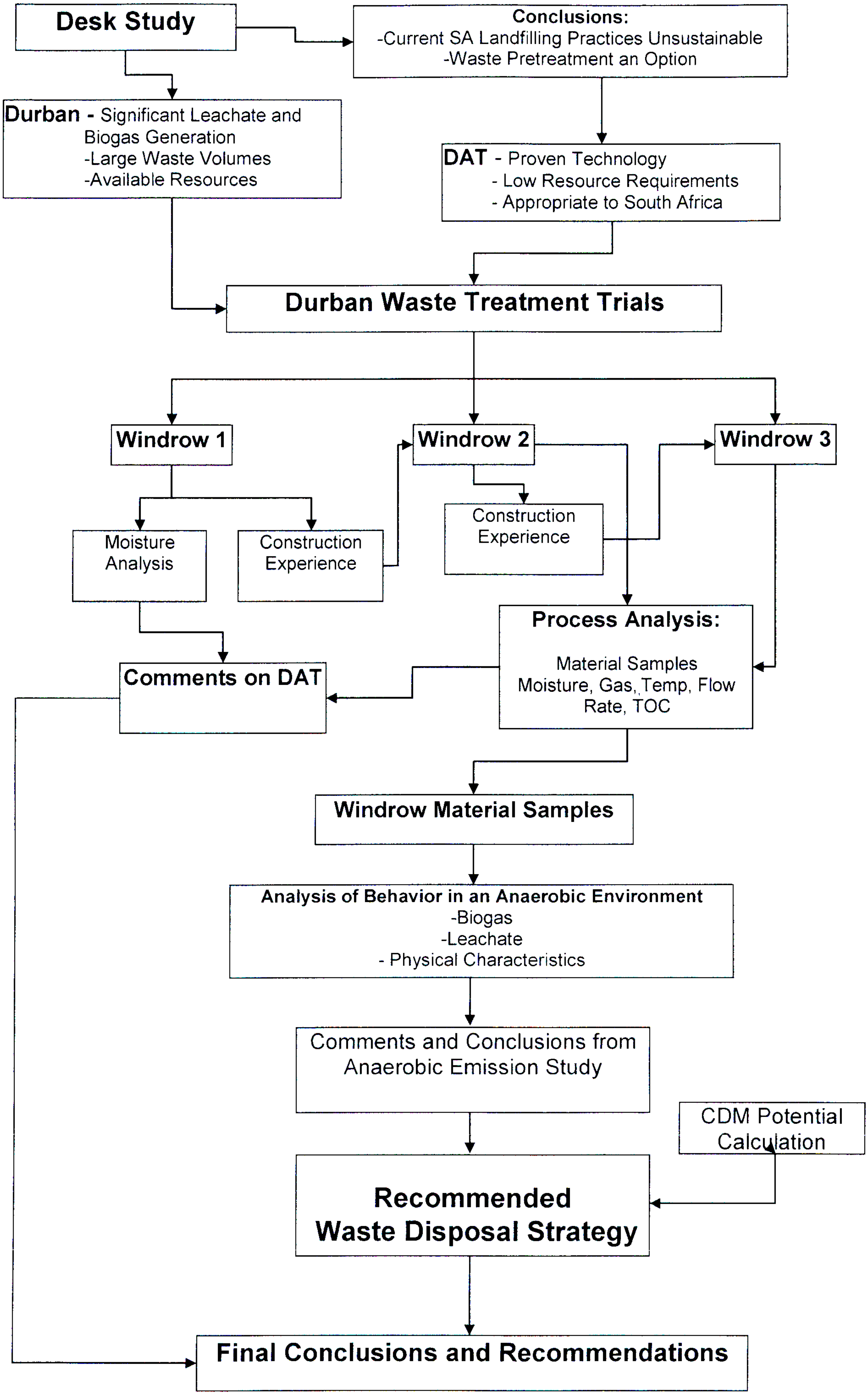

Figure 4.1 Methodological approach of the research

4.1.1 Description of Region: Durban

The city of Durban was selected as the area in which to evaluate the mechanical biological pretreatment of waste using the DAT. Durban, located on the eastern coast of South Africa, is the largest city and producer of solid waste in the province of KwaZulu Natal. The region receives annual average rainfall of 1009mm. The monthly temperatures range from the average maximum of 28 °C in January and February, to the average minimum of 11 °C in June and July. The highest and lowest recorded temperatures are 40°C in October and 3°C in July and August (Table 4.1.).

Due to the warm climate, the landfills rapidly become biologically active, producing significant volumes of landfill gas within a period of 6-9 months (Lombard, 2000; Bowers, 2002). The high rainfall results in the generation of large volumes of leachate. The significant generation of landfill emissions motivates for the inclusion of the mechanical biological treatment step to landfilling operations in the region, and directed the decision to base the research in Durban.

4.1.2 Site: Bisasar Road Landfill Site

The Bisasar Road Landfill Site, established in 1980, is situated approximately eight kilometres north of the Durban central business district. The landfill, operated by Durban Solid Waste (DSW) covers and area of 44 ha, and at the time of writing, the site occupies approximately half of the projected airspace of 21 million cubic meters. The landfill site receives an average of three thousand tons of waste per day with waste compaction density between 1.0 and 1.4t/m^3 achieved (Bowers, 2002). It is classified as a GLB$^+$ landfill site according to the Minimum Requirements for Landfill Design (DWAF, 2005) and is expected to serve the area for a further 4 to 6 years. Originally, the landfill was unlined and operated as an attenuation landfill. The introduction of the Minimum Requirements in 1994 has resulted in the construction of fully lined containment cells. These cells are constructed over the existing landfill and employ full leachate collection. The site generates approximately 250 000 litres of leachate daily (Strachan et. al, 2000) and there is an active gas extraction system in some areas of the landfill (Figure 4.2).

Table 4.1 Durban Climate Information (South African Weather Service)

Month	Highest Recorded (°C)	Average Daily Maximum (°C)	Average Daily Minimum (°C)	Lowest Recorded (°C)	Average Monthly (mm)	Average Number of days with >= 1mm	Highest 24 Hour Rainfall (mm)
January	36	28	21	14	134	15	110
February	34	28	21	13	113	13	197
March	35	28	20	12	120	13	160
April	36	26	17	9	73	9	106
May	34	25	14	5	59	7	111
June	36	23	11	4	28	5	109
July	34	23	11	3	39	5	69
August	36	23	13	3	62	7	91
September	37	23	15	5	73	11	132
October	40	24	17	8	98	15	105
November	34	25	18	10	108	16	94
December	36	27	20	12	102	15	163
Year	40	25	17	3	1009	130	197

Waste Input

The landfill received close to 1 million tons of waste during between July 2006 and June 2007, of which 469 thousand tons was municipal solid waste and 41 thousand tons was garden refuse. A significant volume of rubble and cover material, totalling 410 thousand tons was also received (Table 4.2). The characteristics of the municipal solid waste are given in Table 4.3. Therefore the waste containing biodegradable fractions would require stabilization, the municipal solid waste and the garden refuse, comprises 52% of the total landfill input. The MSW landfilled annually at Bisasar Road contains a significant proportion of putrescible material – approximately 200 000t or 42.5 % of the total MSW input (Table 4.3). The inert material is used as cover material whereby a layer of 100-150mm is spread over the fresh waste at the end of each day (Bowers, 2002).

The machinery utilised on site consists of landfill compactors, and generic earthworks machinery, such as Articulated Dump Trucks (ADTs), front-end loaders, bulldozers, roller compactors, water tanker trucks and an excavator.

Table 4.2 Waste Disposed of at the Bisasar Road Landfill Site July 2006-June 2007*

Waste Type	Mass (tons)
MSW	468 805
Garden Refuse	40 922
Mixed Loads	12 653
Builder's Rubble	93 466
Cover Material	316 897
Other	44 186
Total	**976 911**

*Source: DSW Weighbridge Data

Table 4.3 Characteristics of Durban MSW (DSW Waste Stream Analysis in Bowers, 2002)

Material Type	Percentage	Mass Landfilled (t)
Total	100,0	468 805
Hard Plastics	6,4	30 004
Soft Plastics	11,0	51 569
Glass	7,1	33 285
Tin/Aluminium	6,9	32 348
Cardboard	9,0	42 192
Paper	10,3	48 287
Putrescibles	42,5	199 242
Other	6,8	31 879

The landfill site provided the necessary space and equipment, and also received enough of the various waste fractions (in particular, garden refuse) required for the MBT trials and was therefore considered to be a suitable site for the research.

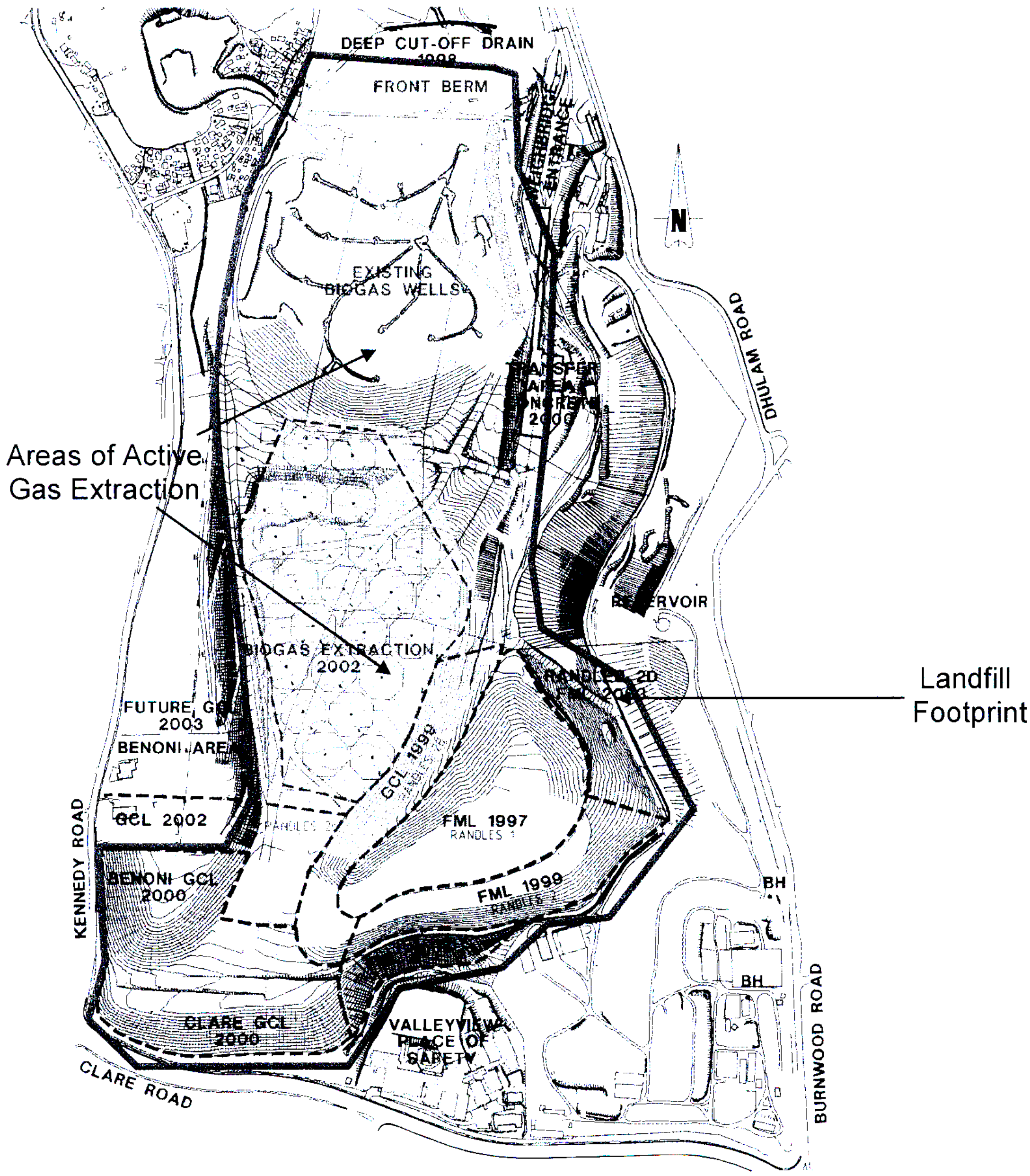

Figure 4.2 Bisasar Road Landfill site showing areas of active gas extraction and new generation lined cells (Courtesy of DSW)

4.2 APPLICATION OF THE DOME AERATION TECHNOLOGY TO DURBAN

It was anticipated that potential problems may arise due to the differences between Germany, where the DAT windrow technology was developed, and Durban, where the windrow trials were to take place. The major concerns were the climate differences, waste stream differences and equipment limitations. These concerns were evaluated in order to avoid potential pitfalls in the full-scale application of DAT in Durban. The optimisation of the DAT technology to the South African conditions was one aspect investigated and developed by this research.

4.2.1 Climatic Difference

As the DAT flow mechanisms rely on the thermal gradient between the ambient air and the windrow dome gas, higher ambient temperatures, as is the case in Durban, may reduce the aeration rates. In order to estimate the effect of ambient temperature on windrow aeration rates, it is necessary to formulate a model that describes the flow mechanisms. The flow velocity of the windrow gas can be calculated using Bernoulli's Energy Equation. Since flow rate is proportional to the flow velocity, this allows for the calculation of the aeration rate. From Figure 3.3, considering point 2 at the base of the dome, and point 0 at the top of the chimney, the energy equation is

$$\frac{p_2}{\rho_2 g} + \frac{v_2^{\,2}}{2g} + z_2 + \frac{\mu^{**}{}_2}{g} = \frac{p_0}{\rho_0 g} + \frac{v_0^{\,2}}{2g} + z_0 + \frac{\mu^{**}{}_0}{g} - h_{wind} + \text{losses} \qquad 4.1$$

where:

at point 0:

p_0 = ambient air pressure

ρ_0 = density of compost gas = ρ_d

v_0 = flow velocty = v

z_0 = height of point 0 above datum = h (Datum is taken as ground level)

$\mu^{**}{}_0$ = intrinsic energy of the gas

h_{wind} = suction on the top of the chimney caused by wind

losses = overall energy losses

at point 2:

$p_2 = p_0 + \rho_d gh + \Delta\rho gh$ (From equation 2.3)

$v_2^{\,2} \approx 0$

$\rho_2 = \rho_d$

$z_2 = 0$

$\mu^{**}{}_2 \approx \mu^{**}{}_0$

Therefore the energy equation can be written as follows:

$$\frac{p_2}{\rho_d g}+\frac{\rho_d gh}{\rho_d g}+\frac{\Delta\rho gh}{\rho_d g}+\frac{\mu^{**}_0}{g}=\frac{p_2}{\rho_d g}+\frac{v_2^2}{2g}+h+\frac{\mu^{**}_0}{g}-h_{wind}+\text{losses}$$

removing like terms and rearranging the equation gives

$$\frac{\Delta\rho h}{\rho_d}+h_{wind}=\frac{v_2^2}{2g}+\text{losses} \qquad 4.2$$

The losses can be consolidated into one empirical loss term:

$$\text{losses}=K\frac{v_2^2}{2g} \qquad 4.3$$

where K is the coefficient of energy loss

Substituting equation 4.3 into 4.2

$$\frac{\Delta\rho h}{\rho_d}+h_{wind}=\frac{v_2^2}{2g}+K\frac{v_2^2}{2g}$$

$$v_2=\sqrt{\frac{2g\left(\frac{\Delta\rho h}{\rho_d}+h_{wind}\right)}{(1+K)}} \qquad 4.4$$

If the suction head due to wind is removed, as is the case for windless conditions, the equation 4.4 becomes

$$v_2=\sqrt{\frac{2g\left(\frac{\Delta\rho h}{\rho_d}\right)}{(1+K)}} \qquad 4.5$$

Therefore, on a windless day, the flow velocity, and thus the flow rate, is proportional to the square root of the density difference ($\Delta\rho$) between the ambient air and the gas inside the windrow dome.

Equation 4.5 can be used to estimate the effect that higher ambient temperatures would have on flow velocity. If the unknown parameter for energy losses (K) is assumed to be constant for all windrows, the flow velocities under a given condition can only be compared to the flow velocities of another given condition. I.e.

$$\frac{v_{T1}}{v_{T2}} = \sqrt{\frac{\left((\frac{M_a}{T_1} - \frac{M_d}{T_d})h \Big/ \rho_{d1}\right)}{\left((\frac{M_a}{T_2} - \frac{M_d}{T_d})h \Big/ \rho_{d2}\right)}} \qquad 4.6$$

where

v_{T1} is the flow velocity for ambient temerature T1

v_{T2} is the flow velocity for ambient temperature T2

The temperature of 0°C is taken as the standard for comparison (v_{T2}) of the expected worst case due to the low temperatures experienced in Cottbus (Paar, 2000). The temperatures compared to this value are 10, 20 and 30 °C, and covers the expected seasonal range of values for the Durban area. The density if a function of the constituents of the compost gas and figures of 15% oxygen and 5% carbon dioxide were used for the calculation (Figure 4.3).

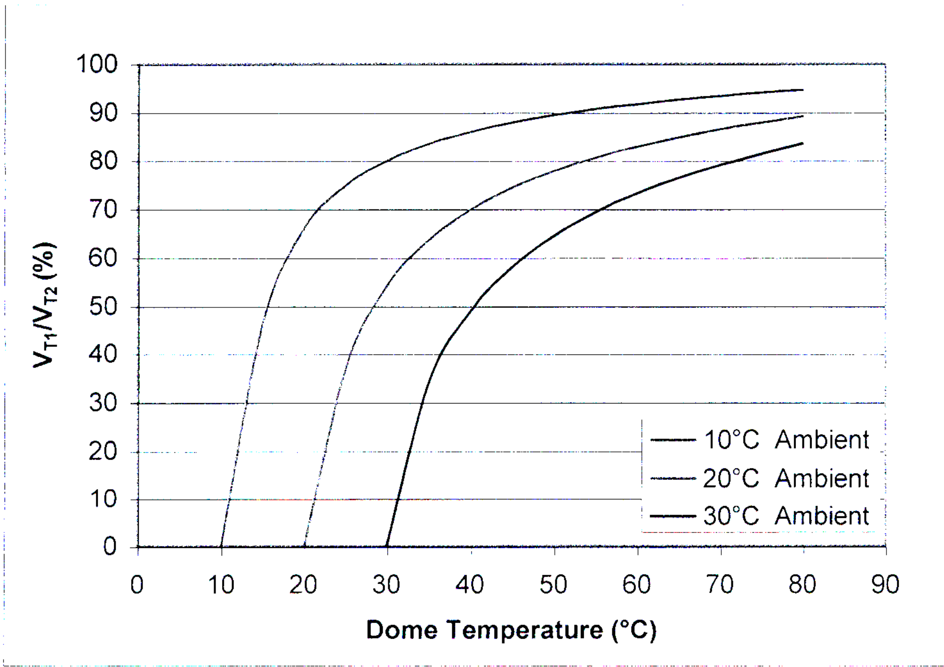

Figure 4.3 The Percentage of Flow Velocity at given temperatures to Flow velocity at 0°C calculated from equation 4.6

From the theoretical results, it is noted that at higher temperature ranges, the relative reduction in flow velocity is the lowest. This range of temperature coincides with the time of most intense biological activity, which requires the highest flow rate. From Figure 3.6, the upper range value of required oxygen concentration (15%) is reached after approximately 15 days at a temperature of 60°C. At 60°C, the flow velocity expected in Durban would be between 10 and 30% less than that expected in Cottbus. Assuming that the windrow temperatures remain above 35°C for the entire composting period as per Figure 3.6, the chimney flow velocity expected in Durban is between 20 and 65% less than that of Cottbus. However, the oxygen content of the Cottbus windrows remains well above the minimum oxygen levels (10-15%) and thus there is a degree of flexibility in terms of lower aeration rates. As the temperatures in Durban may rise as high as 40°C (Table 4.1), this may result in a condition of zero airflow. However, it is anticipated that during these conditions, the oxygen contained in the voids within the windrow will be sufficient for the lower rate biological activity that is expected at windrow temperatures of the order of 35°C, until the ambient temperature falls and the aeration resumes.

Based on the anticipated differences in ambient temperatures the windrow aeration will not be as efficient in Durban as it is in Cottbus. However, due to the high concentration of oxygen measured from the Cottbus windrows, it is likely be sufficient to ensure aerobic conditions for the duration of the composting period, if a conservative approach is adopted with respect to the windrow construction. As the ambient temperatures are not likely to be as high as 30°C for prolonged periods of time, the effect during the early intensive stages is not expected to be as high as 30%, thus an inefficiency of 20% will be used as basis of construction parameters.

4.2.2 Equipment Requirements

The Bisasar Road MBT trial operation lacked some of the necessary equipment for optimal operation. The high capital cost required to provide this equipment was not warranted for this stage of the research, thus motivating the search for suitable alternatives. The mechanical treatment process steps, the equipment utilised for those steps in Cottbus, and the alternatives employed at Bisasar Road are summarised in Table 4.3.

4.2.3 Material Requirements

The waste stream differences between Cottbus and Bisasar Road presented a problem in terms of availability of structural material due to the practice of separate collection in Germany. As an alternative, selected fractions of garden refuse was used as the structural fraction. The garden refuse was selected on the basis of fulfilling the structural component of the windrow input material, and thus relatively thick branches, of the order of 10mm to 200mm in diameter, was the criterion for selection.

Table 4.4 Comparison between mechanical plant

Pretreatment Step	**Cottbus Equipment**	**Bisasar Road Alternative**
Mixing	Crane with Grab	Excavator Pay-loader
Shredding/Crushing	Low Speed Shredder	Landfill Compactor Excavator Passage of Water Truck Roto-Press Refuse Trucks
Wetting	Sprinkler	Water Truck (20 000*l*)

Green and leafy material was rejected, as this would contribute to the more readily degradable fraction of input waste without any structural benefit. The supply of suitable structure material proved to be the limiting factor when accumulating enough materials for windrow construction and operations could only proceed once a suitably large stockpile of structural material had been accumulated. For Windrows 2 and 3, pine bark was used to serve as a bulking agent and a moisture sink for provision of moisture over the composting period. The bark was also used as cover material for all the windrows.

A ratio of one part general waste to one part structural waste (including pine bark), by volume, was used in the construction of all three windrows. Although the general to structural waste ratio used in Cottbus is 2:1, the conservative value of 1:1 was used to provide more voids and thus ensure a completely aerobic windrow.

4.2.4 Windrow Size and Dome Layout

In light of the concerns raised by the climatic differences, the windrows constructed were 20% smaller in cross-section than that of the windrows used in Cottbus. The cross-sectional dimensions of the Bisasar Road Windrows are shown in Figure 4.4. The positions of the domes and chimneys relative to each other remained unchanged at 5m.

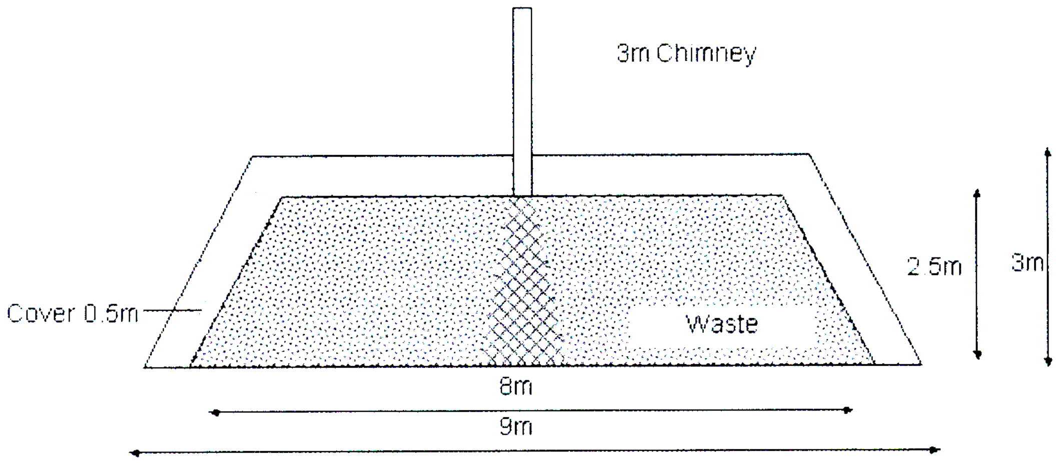

Figure 4.4 Cross Section Dimensions for Durban Windrows

The length of the windrow is directly related to the number of domes used, as the spacing of the domes is a fixed parameter. The windrow needed to be sufficiently large in order to provide representative windrow performance. It was assumed that the two domes on either end would experience airflow patterns that are not representative of the internal domes, which make up the majority of domes in the Cottbus operation. It was therefore decided to construct windrows containing at least three domes with the centre dome assumed to reflect the behaviour of all internal domes. Windrow 1 was constructed using six domes, three domes for waste and three domes for pine bark (Figure 4.5). Windrow 2 was constructed using three domes (Figure 4.6), and Windrow 3 was constructed using four domes, as one would be removed after 8 weeks for the lysimeter trials (Figure 4.7).

4.3 CONSTRUCTION OF THE WINDROWS

Waste Input

As previously discussed, for Windrow 1, one half of the windrow comprised solid waste and structural material in a ratio of 1:1 (Plate 4.1) and the other half was comprised entirely of pine bark. For Windrow 2 and 3, municipal solid waste, structural material and bark were used in a ratio of 2:1:1

Plate 4.1 Windrow input material consisting of structural waste (left) and general waste (right)

Plate 4.2 Rotopress refuse trucks provided suitably shredded municipal solid waste for Windrows 2 and 3.

Characteristics of Input Material

Based on a 2:1:1 ratio of pine bark, the portions of the material are presented in Table 4.4.

Table 4.5 Windrow Input Material

Input Material	Volume (m^3)	Density (kg/m^3)	Mass (kg)	Mass %	Mass in Windrow (kg)***
MSW	0.5	300*	150	50	250
Pine Bark	0.25	400**	100	33.333	166.7
Wood	0.25	200*	50	16.667	83.3
Total	1	400	300	100	500

*Estimated

**Measured

*** Based on average Windrow Density of 500kg/m^3 (Paar, 2000).

Shredding

For Windrow 1, the waste was initially crushed by repeated passes of a landfill compactor (Plate 4.3). For Windrow 2 and 3, the municipal waste was not shredded as the Roto-Press Trucks delivered waste that was suitably shredded (Plate 4.2). The Excavator was used to break up the large fractions of structural material. Repeated passes of the water truck achieved further crushing of the waste during the wetting step (Plate 4.4).

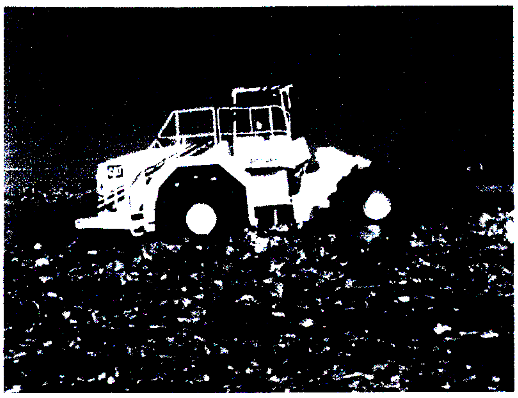

Plate 4.3 Landfill compactor shredding/crushing the waste

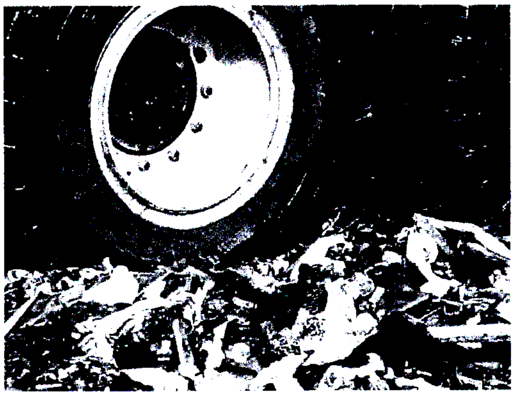

Plate 4.4 The passage of the heavy machinery further crushed the waste

Mixing

For Windrow 1, a front-end-loader created one stockpile of material from separate stockpiles of MSW and SM and than turned the stockpile, mixing the two input materials (Plate 4.5) For Windrow 2 and 3, an excavator was used to load the three different waste streams into an ADT that was sufficiently mixed when it was tipped. This also allowed for better control of waste input ratios (Plate 4.6).

Plate 4.5 Front-End-Loader mixing the waste for Windrow 1

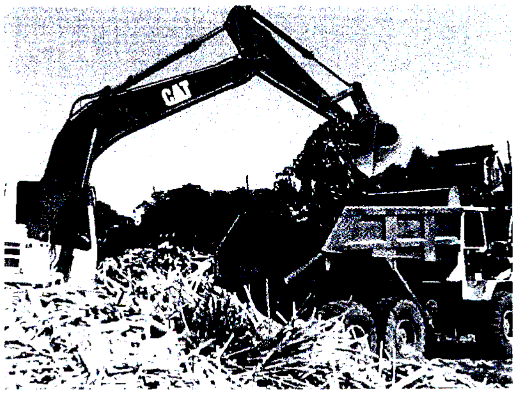

Plate 4.6 An Excavator Loading an ADT during the construction of Windrow 3.
Bulky waste is shown in the foreground

Adjusting the Moisture Content

For all three windrows a water truck was used to wet the waste. For Windrow 1, the water truck sprayed water onto the waste stockpiles (Plate 4.7), while for Windrow 2 and 3 the truck drove directly over the waste that had been spread out over a larger area (Plate 4.8).

The volume of water added to the waste was calculated by estimating the moisture content required to increase the initial value to 55%. For Windrow 2 the estimation for the initial waste moisture content is outlined in Table 4.5.

Table 4.6 Parameters used to calculate input moisture for Windrow 2

Material	Moisture Content (% Wet)	Density (Total) kg/m^3
General Waste	50	300
Structural Material	10	200
Pine Bark	47*	400*
Overall	43	400

*Measured

This necessitated the input of approximately 10 000 litres of water, and thus half a tank (11000 litres) from the 22 000 litre water tanker was sprayed over the waste. However, the trial results indicated that Windrow 2 dried out too early (Refer Chapter 5), resulting in a decrease in biological activity. Therefore a more conservative approach was adopted in the calculation of the moisture content required for Windrow 3. The initial moisture content of the incoming general waste and SM was assumed to be 45% and 0%, respectively. Additionally, a greater volume was allowed for losses during the wetting operation, such as water runoff and evaporation. The calculated requirement was 13000 litres, and therefore a full 22000-litre tank from the water tanker was used. If all the water sprayed onto the waste were absorbed, the waste moisture content would rise to 62%, which is slightly outside of the recommended range of 50-60%. However, a large volume of water loss resulted from runoff, and thus the actual value is more likely to lie within the recommended range.

Plate 4.7 Water Truck Spraying water onto waste stockpile during the construction of Windrow 1

Plate 4.8 The water truck passed over the waste during the construction of Windrow 2

Construction of the Windrow

For Windrow 1 and 3 a front-end loader was used to create the windrow once all the required pretreatment stages had been completed. For Windrow 2 the ADTs dumped the waste material in the correct position, and an excavator was then used shape the windrow.

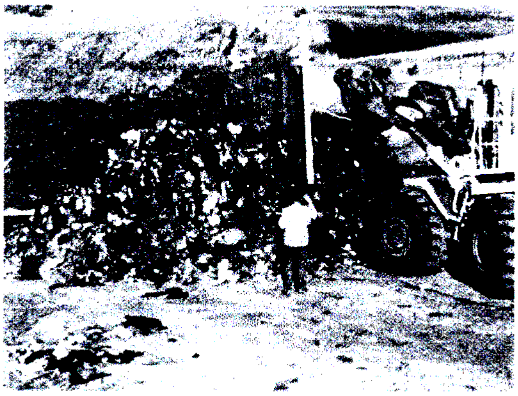

Plate 4.9 The Front-End loader was used to build Windrows 1 and 3

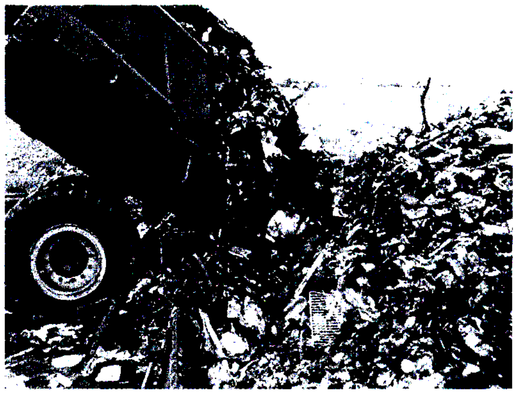

Plate 4.10 The ADT tipped the waste directly into the correct position for the construction of Windrow 2

Cover Material

As there was no composted material available, pine bark was used as a suitable cover material. The cover material was placed using either an excavator (Windrow 1 and 2) or a front-end loader (Windrow 3) (Plate 4.11).

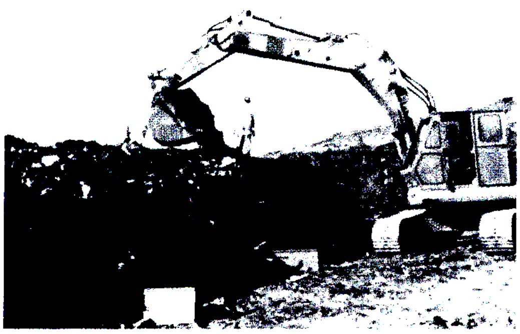

Plate 4.11 Excavator placing Pine-Bark cover on Windrow 1

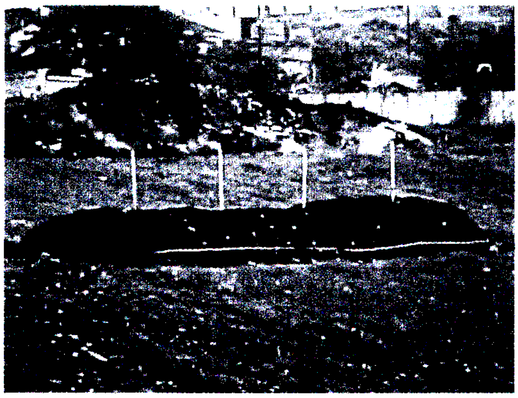

Plate 4.12 The Completed Windrow 3

The results of the mechanical treatment and construction process are summarised in Table 4.7

Table 4.7 Summary of Windrow Construction Parameters

	Windrow 1	**Windrow 2**	**Windrow 3**
Start Date	10-Jun-02	19-Dec-02	28-Mar-03
Waste Input (Ratio)	50% Solid waste, Structural Waste (1:1) 50% Pine Bark	Solid Waste, Structural Waste, Pine Bark (2:1:1)	Solid Waste, Structural Waste, Pine Bark (2:1:1)
Machinery Usage	Landfill Compactor, Front End Loader, Water Truck, Excavator	ADT, Water Truck, Excavator	Front End Loader, ADT, Water Truck, Excavator
Windrow Length (Domes)	3 Domes Waste, 3 Domes Bark	3 Domes	4 Domes
Windrow Volume	$325m^3$ Waste $325m^3$ Bark	$350m^3$	$450m^3$
Time Required	20hrs	20hrs	12hrs

It is noted that the construction of Windrow 3 (Plate 4.12) is the most efficient in terms of windrow volume to construction time.

4.4 ANALYSIS OF THE PROCESS PERFORMANCE

The evaluation of the DAT windrow performance involved the analysis of the following:

Maintenance of a fully aerobic windrow – the oxygen concentration should lie within or above the recommended lower range of 10-15%. Additionally, no significant methane generation should occur.

Thermophilic aerobic activity – the temperature of windrow should rise above 55°C for the purposes of sanitation, and should remain in the range of 35-55°C for the purposes of thermophillic decomposition of the greatest range of materials.

Reduction in the degradable organic content – the ultimate objective the waste treatment operation is the reduction in the degradable organic compounds within the windrow. This can be determined by calculating the cumulative carbon dioxide production from the windrows (Refer Chapter 5).

4.5 ANALYTICAL METHODOLOGY

In order to assess the process performance, the following parameters were analyzed:

Moisture Content

The degradation processes require a moisture content greater than 30-35% for the entire composting period. A moisture content of 55% is recommended for DAT applications. For Windrow 1 Moisture Content Measurements were conducted on a weekly basis by means of thirteen in-situ Time Domain Reflectometer (TDR) Probes (Figure 4.5) commonly used to measure soil moisture content (http://www.sowacs.com/sensors/tdr.html) Windrow 2 and 3, the moisture content was determined by oven drying waste samples at the end of the treatment period.

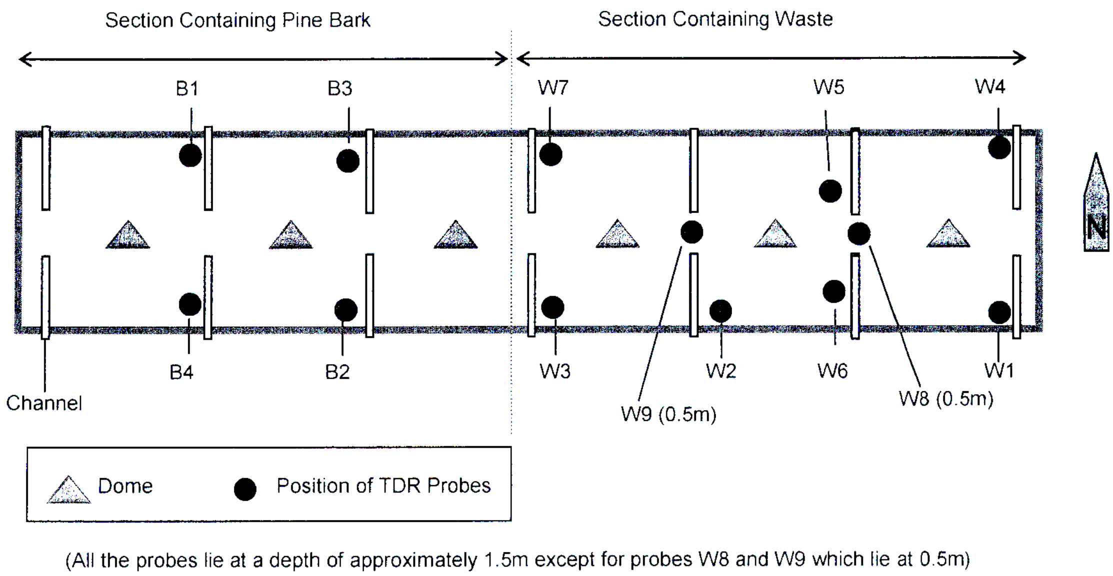

Figure 4.5 TDR Probe Positions for Windrow 1 (Not to Scale)

Temperature and Gas measurements

In order to obtain gas quality and temperature readings from Windrow 2, a removable probe was used. The probe was constructed out of 25mm galvanized steel pipe, with a perforated tip (Plate 4.13). The probe was driven into the windrow at the desired measurement location (Figure 4.6). The location for the placement of probes was chosen on the basis of the theoretical stream flow diagram (Figure 3.5). The anticipated best and worst ventilated areas were investigated, providing the range of values that should be encountered within the windrow. For Windrow 3, 24 fixed probes (nineteen 1m and five 2m probes) also constructed from 25mm galvanized pipe, were installed (Plate 4.14). The probe positions are shown in Figure 4.7. The decision to use a permanent probe was based on the difficulty in obtaining a clear path through the waste during the insertion of the removable probe. However, the fixed probes became severely corroded at a depth of approximately 1m, and consequently only the 1m probes provided reliable readings for the duration of the analysis period as the two-meter probes become perforated halfway along their length. For measurement of the dome exhaust gas and temperature, a hole was drilled into the dome chimney for insertion of the gas probe or thermocouple.

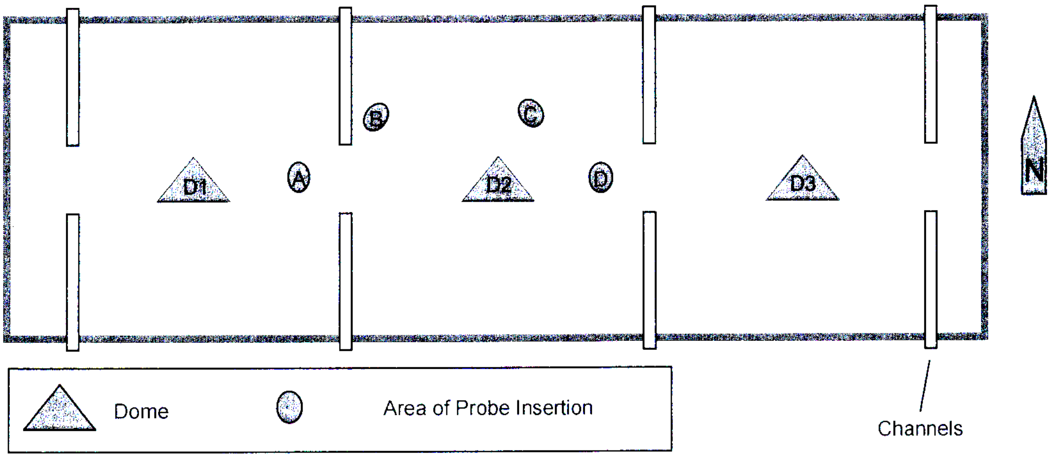

Figure 4.6 Measurement areas for Windrow 2 (Not to Scale)

Windrow Temperature

The windrow temperature must be monitored to ensure that a suitable temperature evolution for effective thermophillic biodegradation. The temperature of the windrow as a whole provides an indication of the degree of biological activity in the windrow, with high temperatures reflecting intensive thermophillic composting, while lower temperatures indicate a lower rate of degradation. The temperature of different areas of the windrow provides an indication of the degree of aeration for that particular area. Finally, the temperature difference between the dome gas and the ambient air is also important, as this is the thermal gradient that drives the aeration process. The difference between dome temperature and ambient temperature also allows for the calculation of the pressure difference that drives the windrow airflow, and allows for the estimation of flow velocities. The windrow body temperature was measured by inserting a thermocouple into the respective probes (Plate 4.13) and the dome temperature was measured by inserting the thermocouple into the hole in the chimney. (Plate 4.15). Ambient temperatures were obtained from weather stations at the Bisasar Road and at the University of KwaZulu Natal approximately 5.5km south of Bisasar Road. This created a more reliable temperature reading, as the influence of the heat that radiates from the windrow would be eliminated.

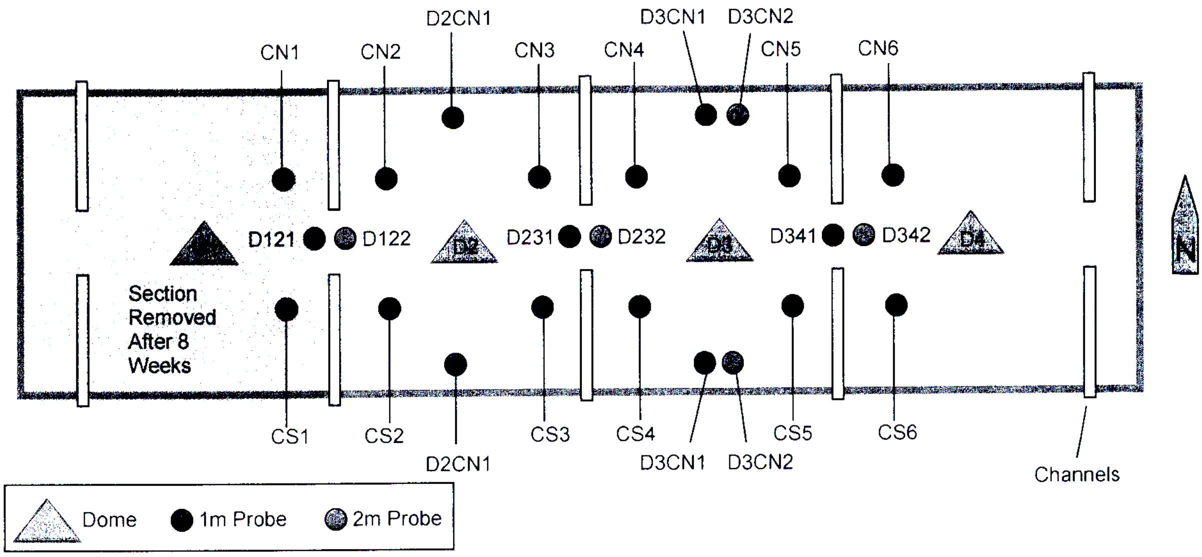

Figure 4.7 Probe Positions for Windrow 3 (Not to Scale)

Gas Concentrations

Oxygen - Measurement of oxygen concentration levels within the windrow is necessary to ensure that the process remains aerobic. Oxygen concentrations that line within or above 10-15% are recommended for the maintenance of aerobic conditions.

Carbon Dioxide – carbon dioxide is produced by the aerobic degradation of organic material within the windrow. Measurement of carbon dioxide, coupled with aeration flow rates, allows for calculation of the quantity of organic material that is degraded.

Methane - Strictly anaerobic microbes produce methane. The presence of methane therefore indicates a *stable* anaerobic environment. This situation is extremely undesirable in terms of aerobic treatment and indicates that there are areas that do no receive any aeration. The presence of methane may also be as a result of the upward migration of gas from the underlying layers of landfilled waste.

For measurement of gas quality, an infrared gas analyser (Geotechnical Instruments GA 94A) was used. Due to the high temperature and moisture content of the gas, the tube was run through cold water to cool and dehumidify the gas. This was done in order to protect the gas analyser, which proved to be sensitive to moisture (Plate 4.16).

Plate 4.13 The Removable Probe with the thermocouple in use

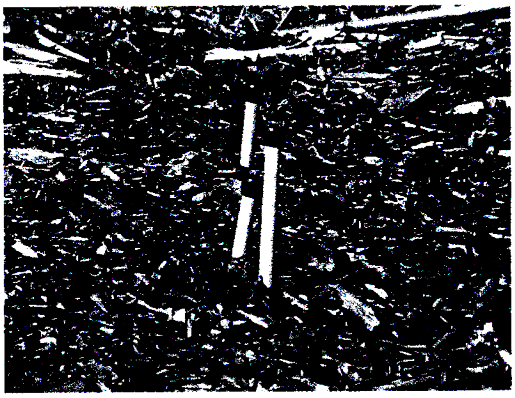

Plate 4.14 The permanent probes for measurement of the Windrow Body

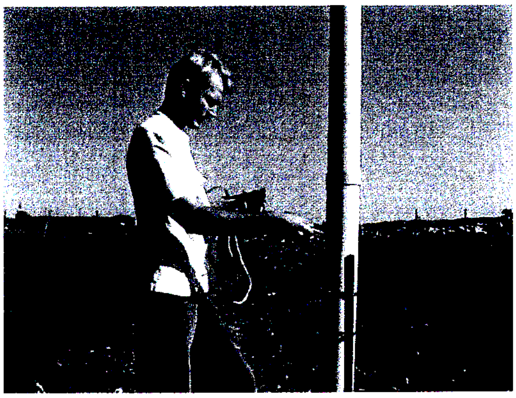

Plate 4.15 Measuring the Dome Temperature

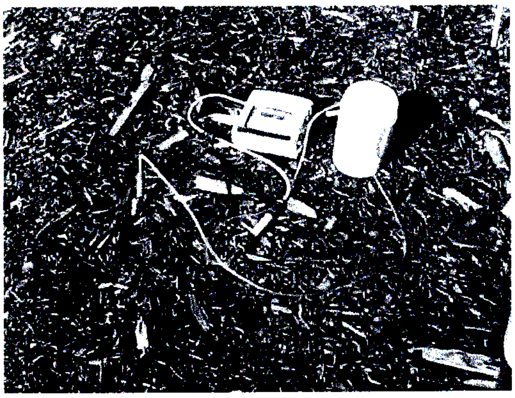

Plate 4.16 Gas measurement from a permanent probe

Dome Chimney Flow Velocity

The velocity of the compost gas as it rises up the chimney allows for the measurement of the flow rate and subsequent calculation of windrow mass balances. The dome flow velocity was measured using a hand-held air-speed meter that was held over the centre of the chimney.

Due to the rapid development of the intensive thermophillic stage, frequent measurements of temperature, gas quality and flow velocity were taken over the first few weeks (every 2-3 days) at mid afternoon. During the relatively stable cooling and maturation stages, the frequency of measurements was decreased (once every 1-2 weeks).

Cumulative Carbon Dioxide Production

Disregarding the synthesis of biomass (found to be relatively negligible), the material balance of substances that are catabolically converted is as follows (Paar, 2000).

$$C_aH_bO_cN_d + vO_2 \rightarrow xCO_2 + yH_2O + zNH_3 \quad 4.7$$

Calculating the organic carbon ($orgC_i$) for each particular organic molecule (i) therefore enables the estimation of carbon dioxide produced during the degradation reaction. The mass of organic carbon available is calculated as follows:

$$orgC_i = \frac{a * MM_C}{MMi} = \frac{a * MM_C}{aMM_C + bMM_H + cMM_O + MM_N} \quad 4.8$$

where

MM_C is the molar mass of carbon = 12g/mol

MM_H is the molar mass of hydrogen = 1g/mol

MM_O is the molar mass of oxygen = 16g/mol

MM_N is the molar mass of nitrogen = 14g/mol

thus equation 4.8 becomes

$$orgC_i = \frac{a * 12}{a * 12 + b * 1 + c * 16 + d * 14} \quad 4.9$$

The chemical makeup of the organic material has been approximated by various studies, the values of which are presented in Table 4.8.

Table 4.8 Chemical Formulae for the Organic Content of Various Waste Materials

Material	Formula	Source
Food Wastes	$C_{18}H_{26}O_{10}N$	Kayhanian et. al (1992) in Haug (1993)
Wood	$C_{295}H_{420}O_{186}N$	Corey in Haug (1969)
Mixed Paper	$C_{266}H_{434}O_{210}N$	Kayhanian et. al (1992) in Haug (1993)
Pine Bark	$C_{295}H_{420}O_{186}N$	Haug (1993)

Biodegradable Organic Fraction

Not all of the organic content is biodegradable due to the presence of resistant molecules, in particular lignin (Haug, 1993). A factor known as the biodegradable organic fraction (f_{bi}) is applied to the organic carbon content ($orgC_i$) to obtain the total biodegradable organic carbon ($orgCb_i$) (Cossu et. al. 1996). Thus the biodegradable organic carbon is as follows:

$$orgCb_i = orgC_i \cdot f_{bi} \cdot (1 - w\%) \cdot m_i \qquad 4.10$$

where:

f_{bi} is the biodegradable organic carbon factor

$w\%$ is the moisture content of the material i

m_i is the total mass of the material i

Based on the information in Tables 4.3 and 4.5, the $orgCb_i$ for the windrow input material calculated. The results are presented in Table 4.9

Table 4.9 Biodegradable Organic Carbon of Windrow Material Input.

Material	Mass per m^3 Windrow Material	Moisture content w%*	Dry Mass DM	orgC / kg DM	f_{bi}*	orgCb per m^3 Windrow Material
Unit	kg	%	kg	kg		kg
Putrescibles (Food Waste)	106,25	60	42,5	0,42	0,8	14,12
Mixed Paper	48,25	8	44,4	0,25	0,5	5,65
Wood	83,3	20	66,7	0,23	0,5	7,60
Pine Bark	166,7	47	88,3	0,13	0,25**	2,81
Total						**30,19**

*Source: Cossu et. al. (1996)

** Estimate based on 60 day soil incubation test in Haug (1993)

Thus for every cubic meter of windrow material, there is 30.2kg of degradable organic carbon. The overall orgC for the windrow material is estimated to be 367mg/g (DM).

In order to assess the degree of degradation, the total mass of carbon dioxide released is calculated. This is then compared to the actual measured carbon dioxide released (Chapter 5).

One mole of organic carbon generates 1 mole of carbon dioxide. The molar volume of carbon dioxide at STP is 22,4, thus the theoretical total carbon dioxide release is:

$$CO_{2prod} = n_{co2} \cdot 22,4 = \frac{30,19}{12} * 22,4 = 56,35m^3CO_2 / m^3 windrow$$

The windrow cross sectional area is 17,5m^2 and the length of windrow per dome is 5m, therefore the ultimate volume of carbon dioxide produced per dome is:

$$CO_{2prod} / dome = 17,5 * 5 * 56,35 = 4930,7m^3$$

This figure represents the theoretical maximum carbon dioxide production. However, a lower value is expected due to due to the diffusion of some gases out through the sides of the windrow and the slow degradability of certain materials, particularly the pine bark and wood wastes used in the Durban windrow trials which will not be fully degraded after the composting period of 16-20 weeks.

The results of the windrow analysis are presented in the following chapter.

CHAPTER 5

DOME AERATION WINDROW RESULTS

5.1. INTRODUCTION

The results for each windrow trial are presented and measured against the recommended parameters that are required by conventional aerobic composting systems. The results from the Durban DAT trials are also compared to those of the Plauen DAT trials (Paar, 2000). The recorded values can be found in Appendix A.

5.2. MOISTURE CONTENT

Windrow 1

Moisture content readings were logged using in-situ TDR probes over a period of 97 days, at weekly intervals. The moisture content values ranged between 0 and 30%. The disadvantage of using TDR probes is the relatively small area that is measured in a heterogeneous material such as waste. For this reason, the moisture content is evaluated relative to the initial value measured after wetting of the material. The evolution of the windrow moisture content is shown for the waste portion of the windrow in Figure 5.1 and the bark portion in Figure 5.2. For probe positions, refer to Figure 4.6.

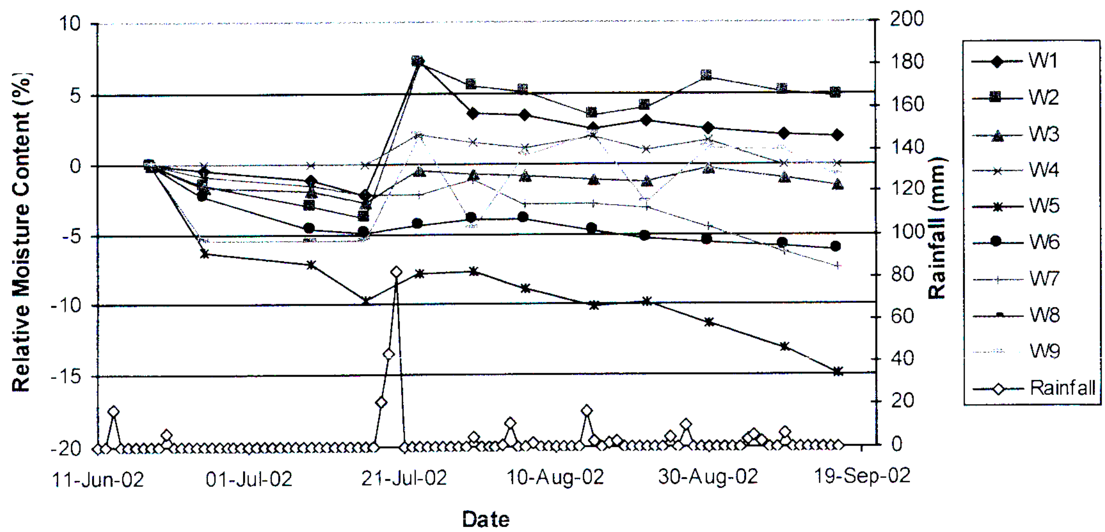

Figure 5.1 Relative Moisture Content for the Waste Section in Windrow 1

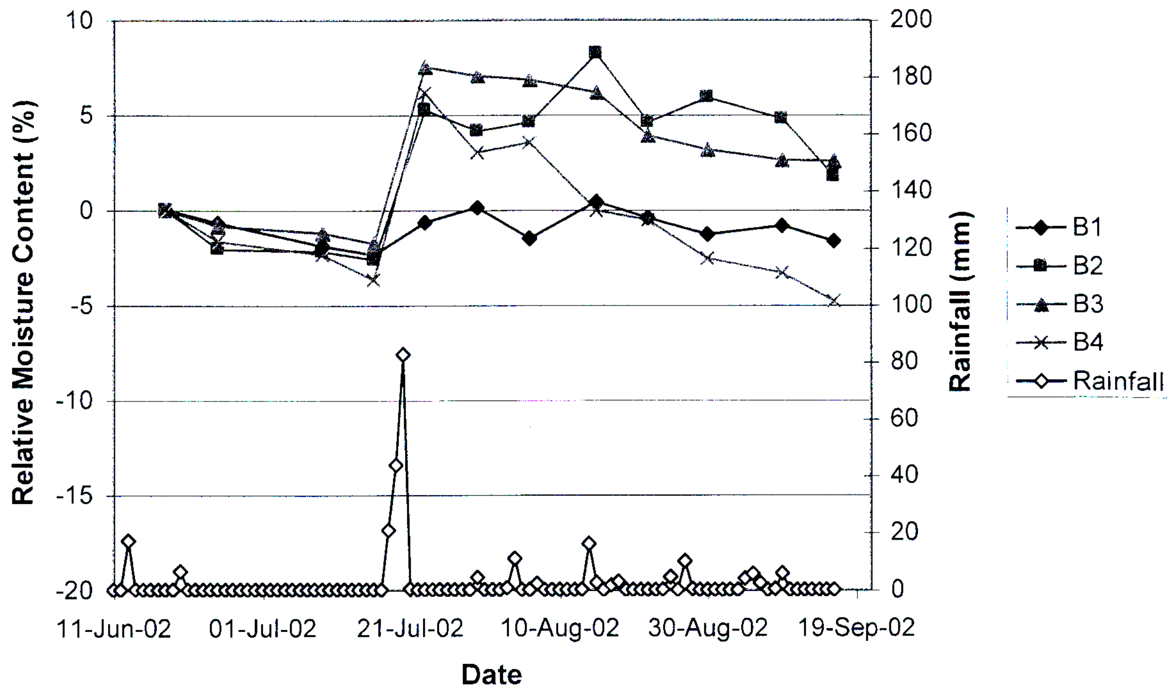

Figure 5.2 Relative Moisture Content for the Bark Section in Windrow 1

The moisture content of the waste section tends to decrease with time, with localised increases due to large rainfall events. The effect of the rainfall is different for each probe, with some probes showing little response to rainfall, while others are greatly influenced. The varying distribution of moisture as a result of rainfall, indicates that the addition of water after the windrow has been constructed is not a reliable method of providing moisture for the entire windrow. It is therefore crucial to provide sufficient moisture for the duration of the composting period as confirmed by the trials in Germany (Paar et al, 1999). For the bark section of the windrow, the effect of rainfall was more uniform than the waste section, due to the homogeneity of the bark. However, the presence of preferential pathways through the bark is still evident in the varying degrees of rainfall infiltration.

Windrow 2 and Windrow 3

The rainfall recorded at the University of KwaZulu Natal's weather station, (approximately 4km away) during the composting period of Windrow 2 and 3 is shown in figures 5.3 and 5.4 respectively.

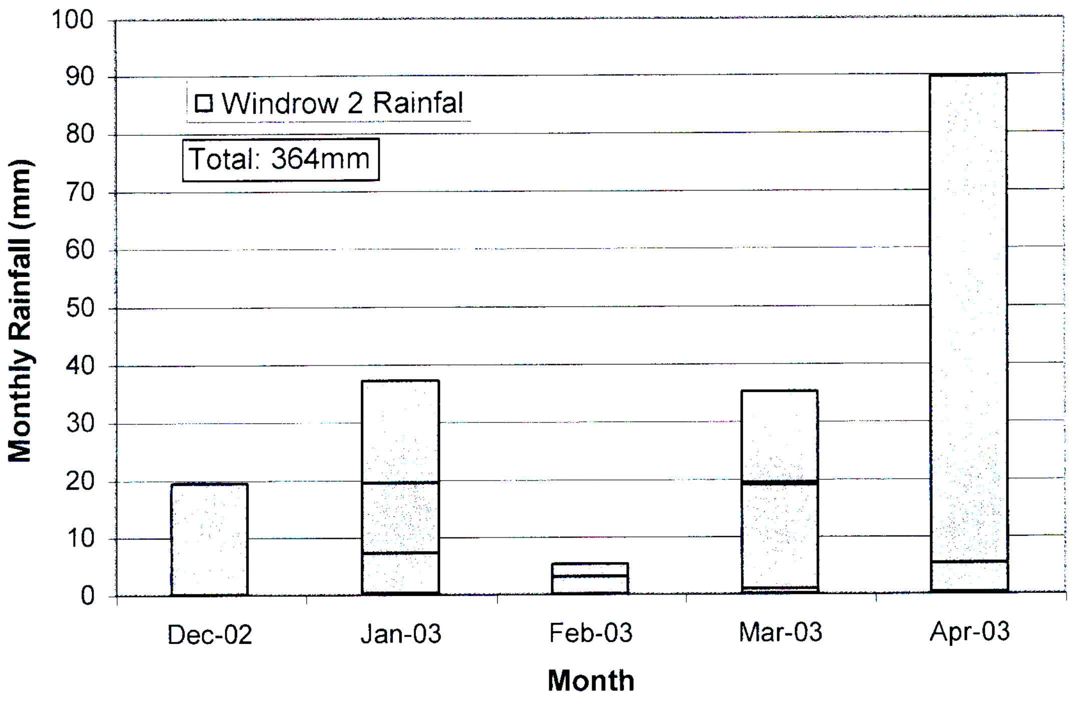

Figure 5.3 Monthly Rainfall During Windrow 2 Composting Period

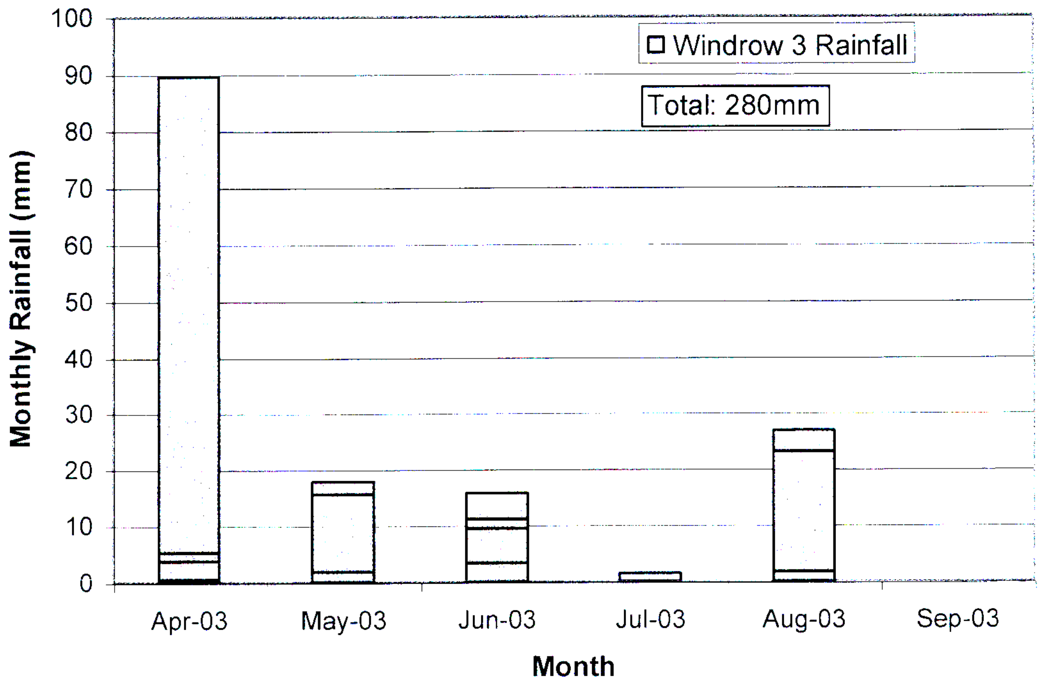

Figure 5.4 Monthly Rainfall During Windrow 3 Composting Period

The moisture contents of the material from Windrow 2 and Windrow 3 were measured in the laboratory and the results are presented in terms of the percentage moisture of the total (wet) weight in Table 5.1.

Table 5.1 Moisture Content of Windrow 2 and 3

Moisture Content:	Initial	8 weeks	Final
Windrow 2	-	-	21% (20 weeks)
Windrow 3	49%	38%	31% (22 Weeks)

The moisture content of 21% for Windrow 2 is well below the minimum recommended moisture content of 30-35%. This suggests that the rate of biodegradation may have slowed down significantly during the composting period, and is shown by an early decrease in biological activity, reflected by a drop in temperature and carbon dioxide concentrations discussed later. The initial moisture content for Windrow 3 is slightly lower than the recommended 55% for the start of the composting stage. After 8 weeks, the moisture had decreased by 11%. The subsequent 14 weeks resulted in a decrease of only 7%. This is as expected as the higher temperatures of the earlier stages result in the removal of more moisture from the windrow. The final moisture content of 31% is equivalent to the average value (30%) obtained from the Cottbus trials (Mollekopf et al, 2002). There was therefore sufficient moisture available for biodegradation for the entire composting period.

Although there was a higher rainfall during the composting period of Windrow 2 (Figure 5.3), than for Windrow 3 (Figure 5.4), the final moisture content of Windrow 2 was still lower than that of Windrow 3. This confirms that addition of moisture once the windrow has been constructed is reliable in providing moisture for the composting process.

5.3. TEMPERATURE

Windrow 2

The results for the dome and windrow body temperatures are presented in figures 5.5 and 5.6. For the dome and probe positions refer to Figure 4.6.

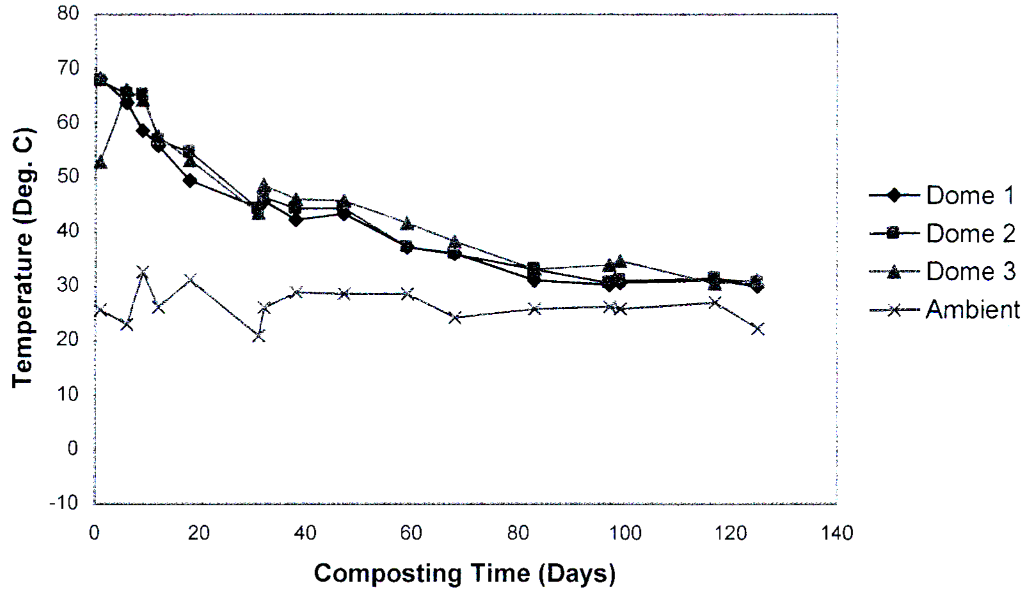

Figure 5.5 Windrow 2 Dome Temperatures

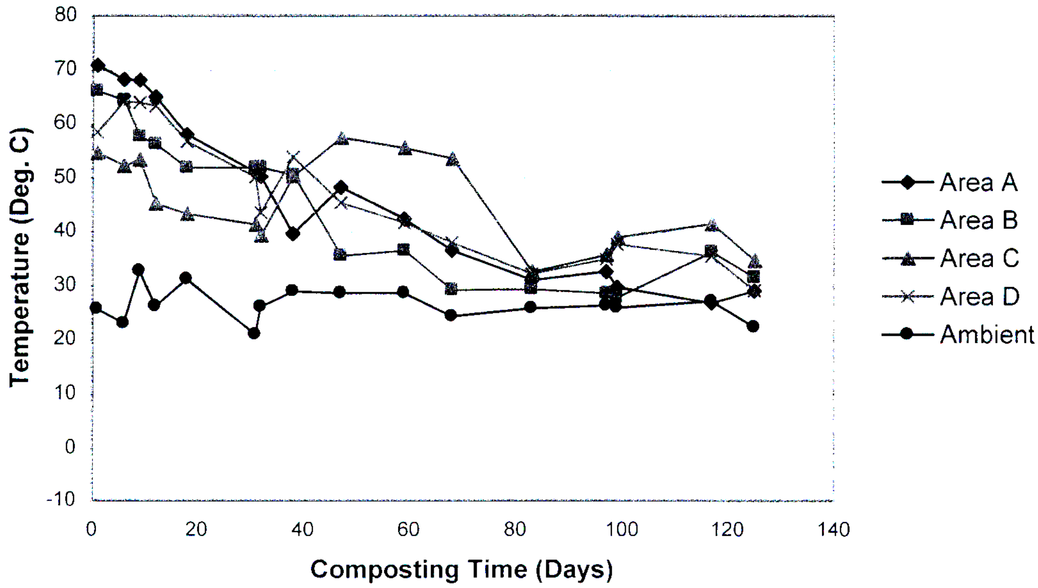

Figure 5.6 Windrow 2 Body Temperatures

The windrows demonstrate the characteristic progression to thermophilic conditions with temperatures peaking at 68°C before entering the cooling and then maturation stages typical of compost systems. From day 80-100 the temperatures are approximately equivalent to the ambient temperatures, indicating a lower rate of biological activity within the windrow.

The coolest areas are the better-ventilated areas (Probes B and C, Figure 4.6) however, after 40 days, the well-ventilated area (Probe C) is consistently the warmest area of the windrow. This is likely to be as a result of heterogeneity of the windrow body, as temperature and gas measurements conducted in one instance 1m apart yielded different results. As for the domes, the coolest areas of the windrow approach ambient temperatures after approximately 80 days of composting.

Windrow 3

The dome temperatures from Windrow 3 are presented in Figure 5.7, and the Windrow body temperatures are presented in Figure 5.8. For the dome and probe positions refer to Figure 4.7. Due to the large number of data points, only the highest, lowest and average values for the windrow body measurements are presented in Figure 5.8 (All the values can be viewed in Appendix A). Note that an area of the windrow, including Dome 1 and probes CN1, CS1 and D12 1 was removed for the lysimeter investigation after 8 weeks of composting.

The windrow dome temperatures rise steeply to 70°C over the first 5-10 days, and then gradually decrease. There is a secondary peak of 70°C after 29 days of composting, although this appears to be discontinuous and is likely to be due to an environmental factor such as wind. The dome temperatures all remain within 10°C of each other for the majority of the composting period, with the temperatures of the two inner domes (Dome 2 and Dome 3) consistently higher than those of the two outer domes (Dome 1 and Dome 4). This is expected as air is drawn through more composting material at the internal domes than at the external domes.

The variation of temperature in the windrow body is far greater and in some areas of the windrow body temperatures remain relatively high after the first week of intensive biodegradation. Temperatures of above 80°C were sustained for extended durations, and only fell below these levels after 86 days of composting. These high temperatures are undesirable as the range of microorganisms that are able to function at these high temperatures is limited. At the other end of the scale, some areas of the windrow body remained quite low and may not have experienced the same level of treatment at the rest of the windrow.

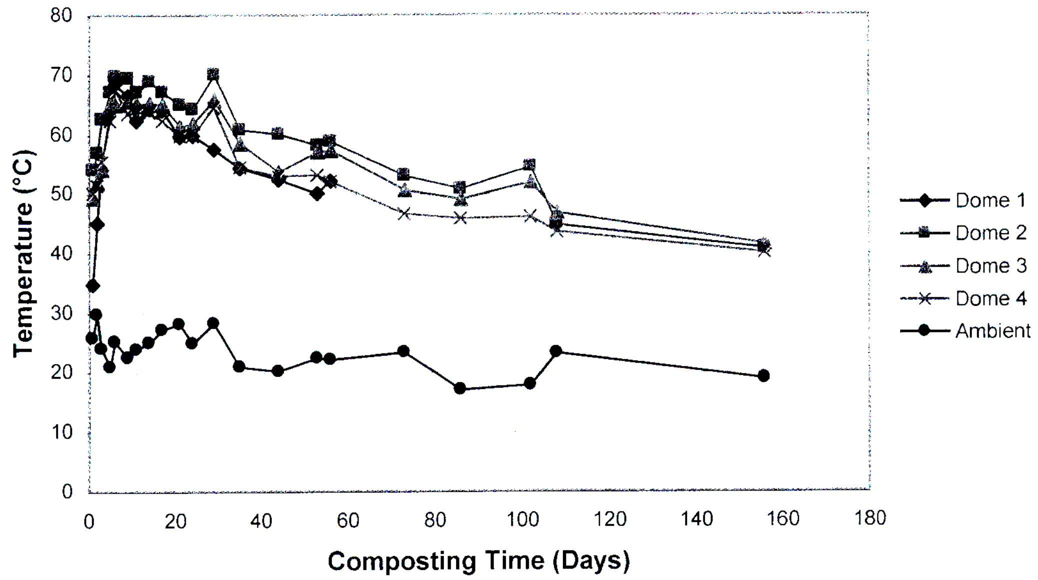

Figure 5.7 Windrow 3 Dome Temperatures

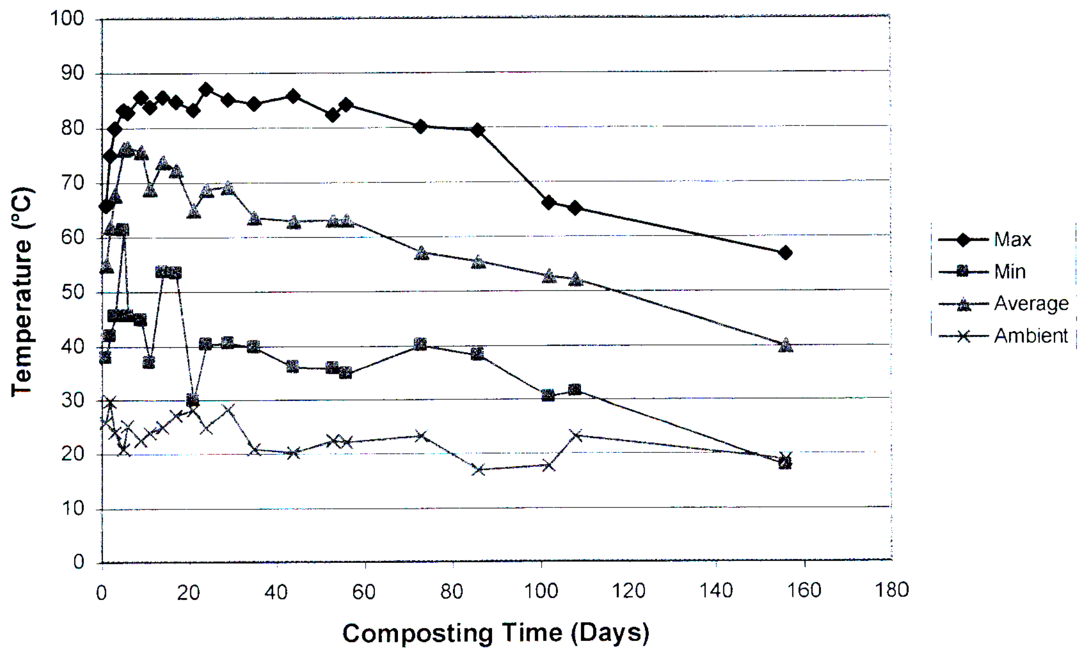

Figure 5.8 Windrow 3 Body Temperatures

Comparisons

The dome temperatures for Windrow 2, Windrow 3, and Plauen Windrows are shown in Figure 5.9.

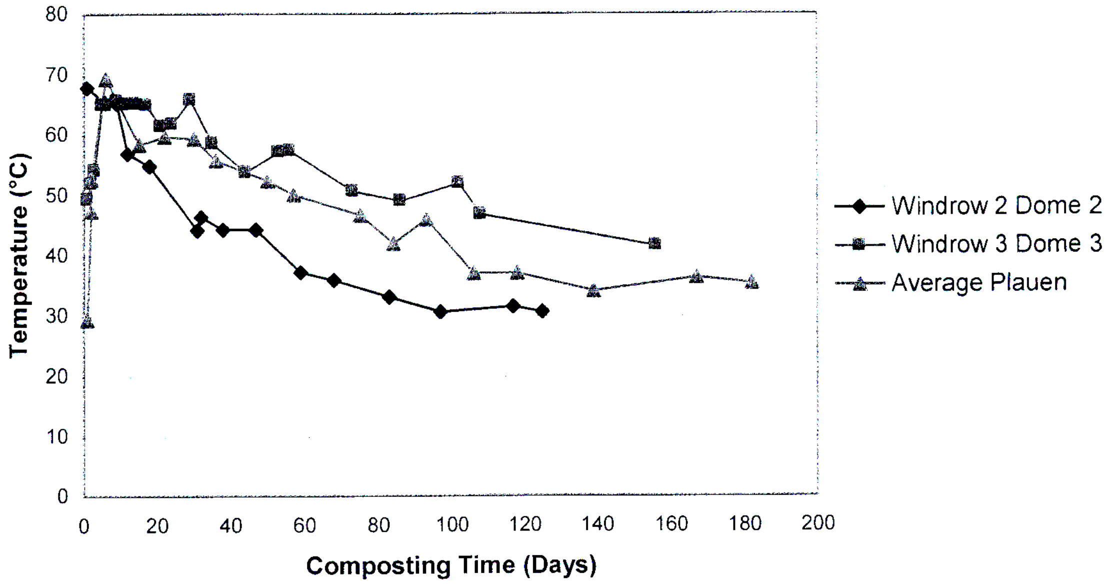

Figure 5.9 Comparative Windrow Temperatures

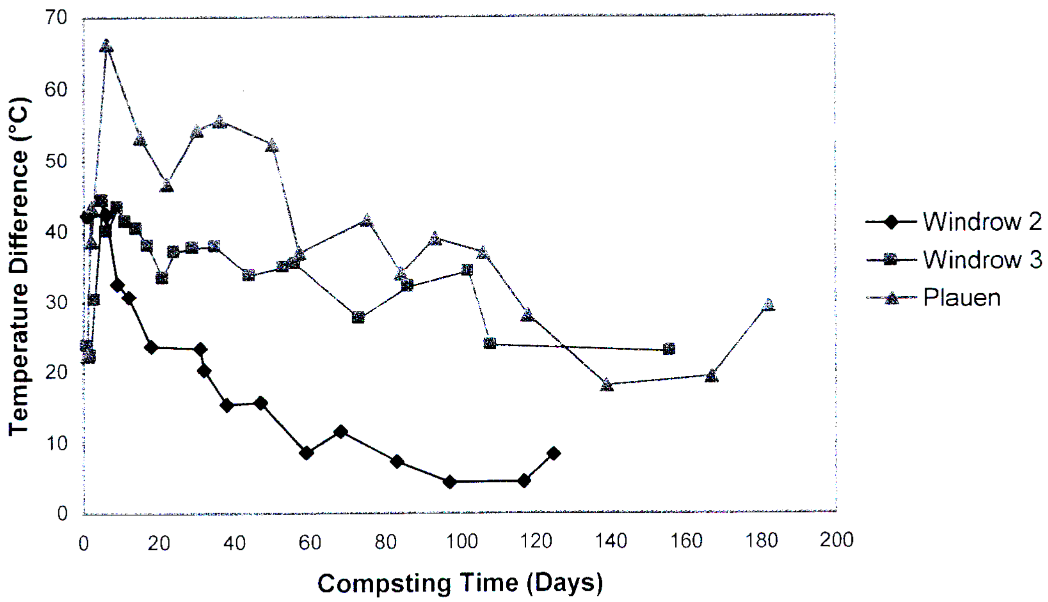

Figure 5.10 Temperature Difference between Dome Gas and Ambient Air

All three windrows reach approximately the same thermophilic peak in similar timeframes. However the temperature of Windrow 2 decreases more rapidly than Windrow 3 and the Plauen Windrows. The lower temperatures of Windrow 2 are significant in terms of the temperature difference that drives the aeration process, as the ambient temperatures for Windrow 2 are the highest, and therefore the temperature difference between the ambient air and dome gas is very low (Figure 5.10). This is expected to negatively impact the rate of aeration. The result from Windrow 3 shows a larger temperature difference and therefore a better rate of aeration is expected to have occurred. The peak temperature difference between the ambient air and the windrow dome of approximately 45°C for the Durban Windrows is 20° less than that of the Plauen Windrows. This would result in a relatively lower rate of aeration for the Durban Windrows. The significance of this lower rate of aeration will be evident in the levels of oxygen in the exhaust gas, and possible presence of methane from anaerobic niches.

5.4. WINDROW GAS CONCENTRATIONS

Windrow 2

The readings of oxygen concentration measured from the domes and from the windrow body are presented in figures 5.11 and 5.12 respectively. The concentration of methane that was measured throughout the duration of the composting period is presented in Figure 5.11.

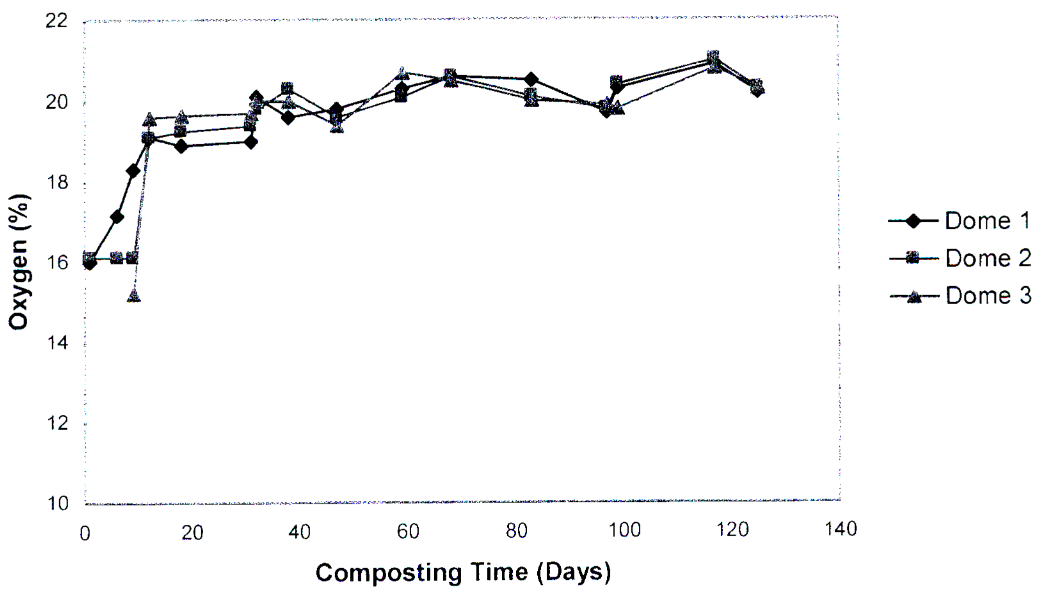

Figure 5.11 Windrow 2 Dome Oxygen Concentrations

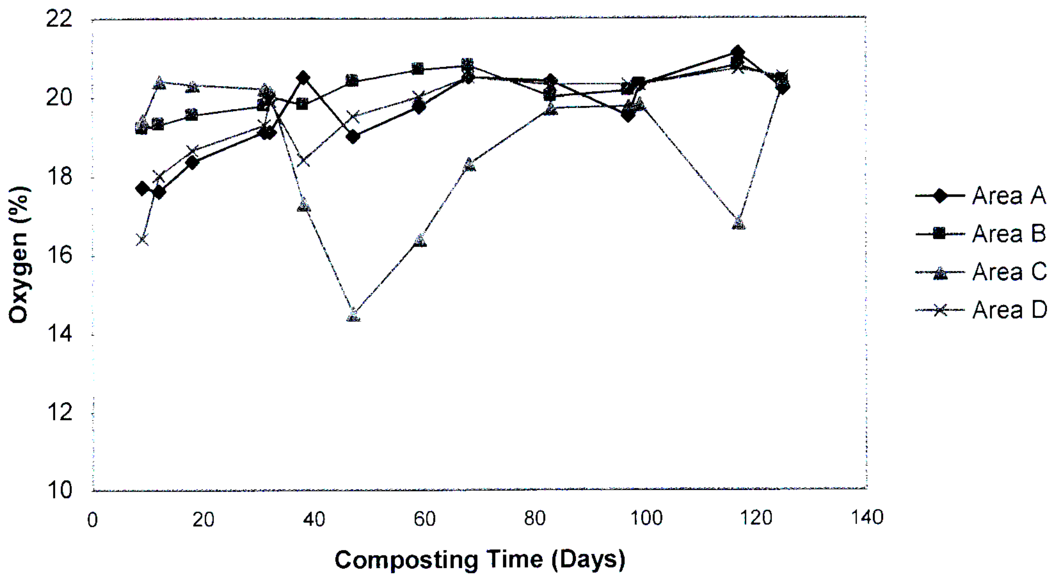

Figure 5.12 Windrow 2 Body Oxygen Concentrations

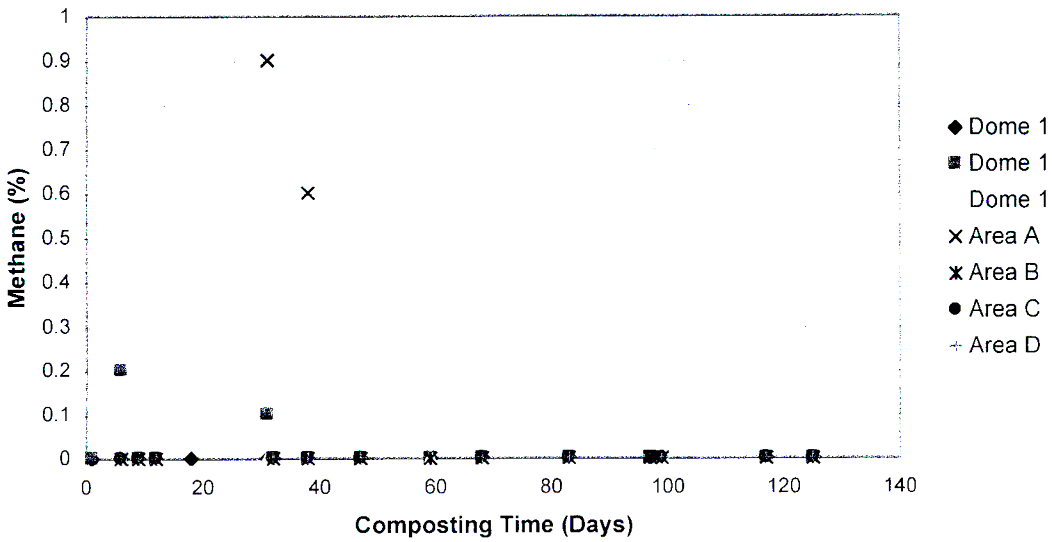

Figure 5.13 Windrow 2 Methane Concentrations

The oxygen concentration in the dome exhaust gas (Figure 5.11) increases rapidly over the first week of composting, rising above the recommended minimum concentration of 10-15% after 9 days of composting and remaining above 15% for the entire composting period. The body of the windrow (Figure 5.12) also maintained oxygen levels above the required concentrations, with the lowest concentrations, measured from Area C, never falling below 10%.

Figure 5.13 shows four instances of methane detection throughout the composting period. The highest concentration of 0.9% occurred after 31 days and overall the presence of methane can be assumed to be negligible.

Both the high concentration of oxygen and infrequent instances of methane detection indicate that the windrow, as a whole, remained fully aerobic for the entire composting period.

Windrow 3

The oxygen concentrations or Windrow 3, measured from the domes and from the probes in the windrow body, are presented in Figure 5.14 and 5.15 respectively. Due to the large number of probes, only the maximum and minimum values and the average of all the probe readings for each respective analysis day are presented for the windrow body. All the measurements can be viewed in Appendix A. The maximum and average values of methane gas detected in the windrow dome and windrow body during the analysis period are shown in Figure 5.16

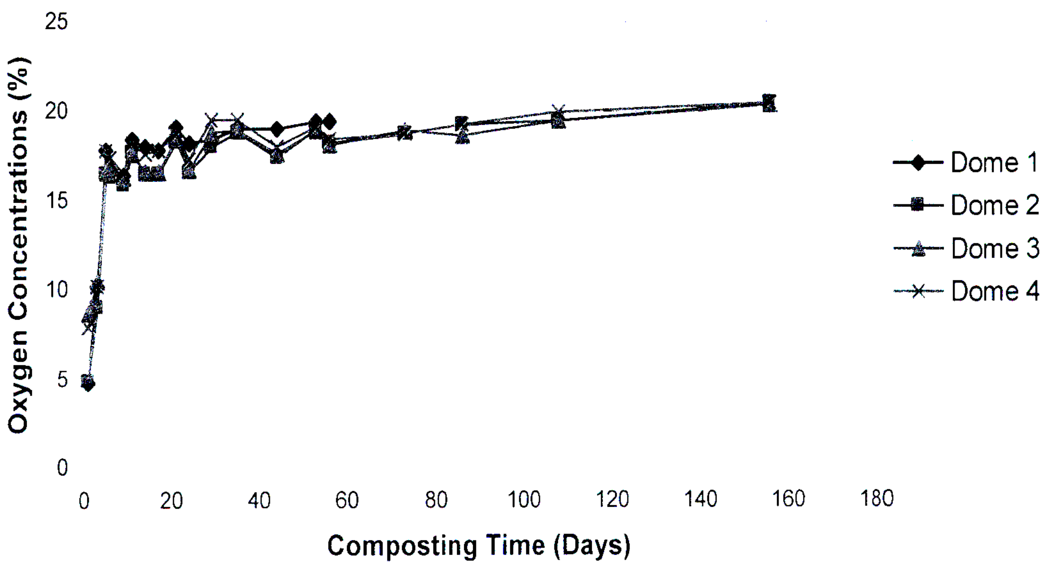

Figure 5.14 Windrow 3 Dome Oxygen Concentration

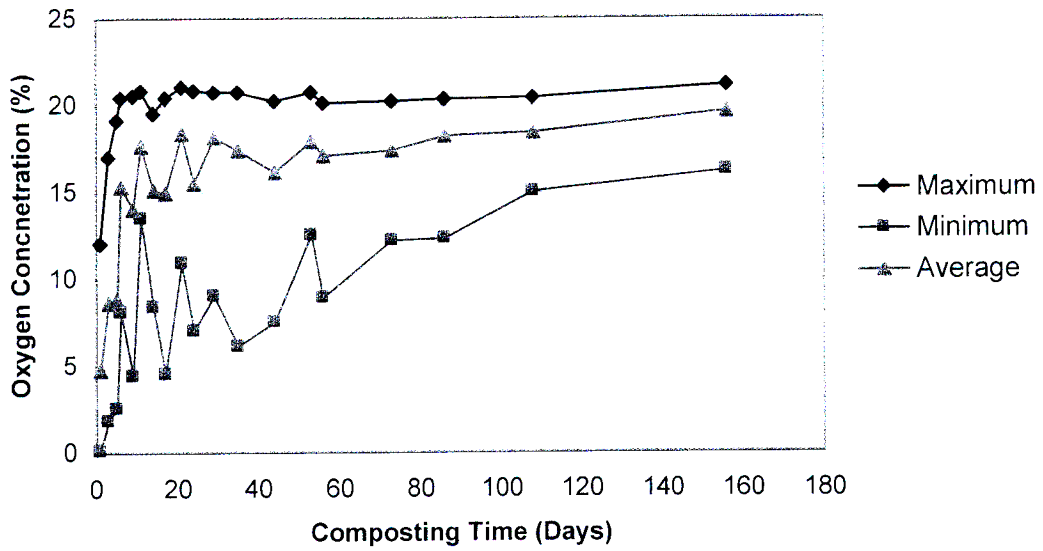

Figure 5.15 Windrow 3 Body Oxygen Concentrations

From Figure 5.14, the optimum oxygen levels (15%) are measured from the dome exhaust gas within 5 days of composting. After which the oxygen levels remain above 15% for the entire composting period. The results from the windrow body (Figure 5.15) show that the average oxygen concentration levels reach 15% after six days of composting, while some areas remain at low levels for a long period of time. It is only after 60 days of composting that all areas reach oxygen concentration levels above 10%, and after 108 days oxygen concentrations greater than 15% are measured from all probes. This variation demonstrates that certain areas receive significantly more aeration than others. In Chapter 3 it was stated that the windrow contains areas of better and poorer aeration (Figure 3.5). The oxygen concentrations in the well ventilated (WV) and poorly ventilated (PV) areas are shown in Figure 5.16. This figure shows that although in the initial stages of composting the well ventilated areas did exhibit slightly higher oxygen concentrations overall, there is no significant difference between the areas that were predicted to be poor or well ventilated. The area most frequently measured as having the lowest oxygen concentration was an area that was predicted to be well ventilated. As the majority of the windrow receives ample aeration, this situation is likely to result from localised areas of poor pore structure. The causes of this may be the heterogeneity of the windrow material, insufficient structural material, localised compaction during construction, settling during the composting stage, or elevated moisture content.

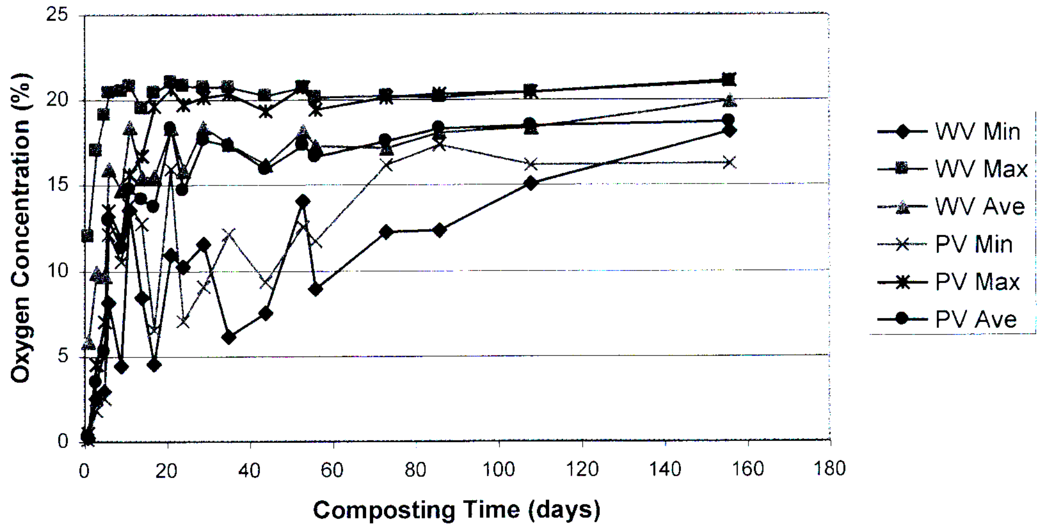

Figure 5.16 A Comparison between the Well Ventilated (WV) areas and Poorly Ventilated Areas (PV)

When considering the windrow methane content (Figure 5.17), it is noted that frequent detection of trace (less than 1%) amounts of methane gas occurred during the analysis period. The average amount of methane detected in the dome exhaust gas is equivalent to the average amounts recorded from the windrow probes. The high incidence of methane detection suggests the presence of stable anaerobic areas within the windrow, and the low oxygen content detected in the windrow body supports this. However, the unusual aspect regarding the presence of methane is that the highest concentrations (0.8%) are detected on the first day of composting. This would imply that stable anaerobic zones were present in the compost material at the time of placement. This is unlikely due to the intensive mechanical treatment stage that would have exposed the sensitive methanogenic bacteria to oxygen. The other potential source of methane is the underlying layers of landfilled waste, which the windrow was built on top of. The same phenomenon occurred during a windrow trial by Polster, (2003), where the study was constructed on the same area as Windrow 3.

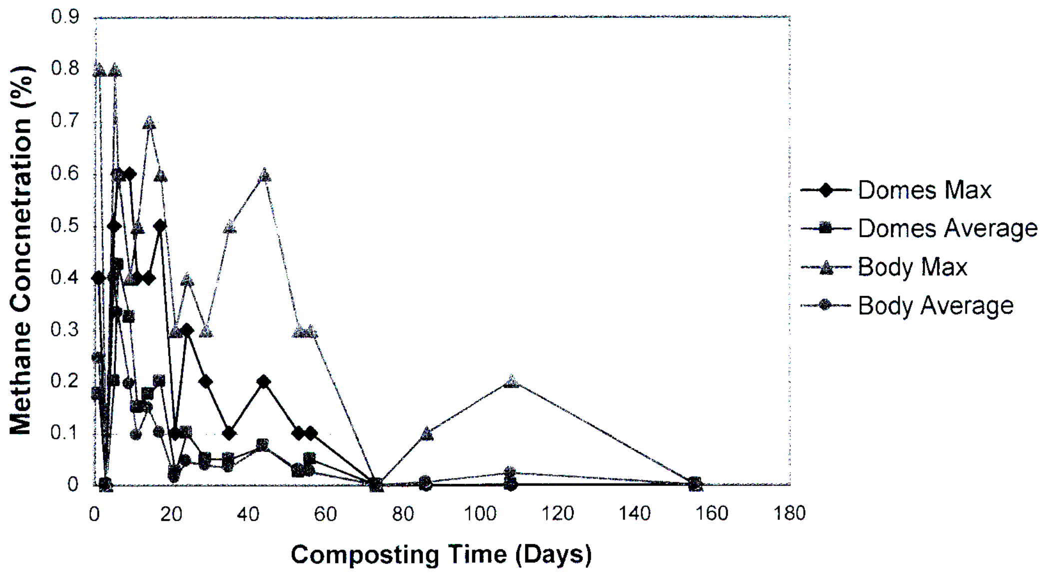

Figure 5.17 Methane Detected in Windrow 3

With respect to the occurrence of methane in the area of worst recorded oxygen concentrations (Probe CN2), the oxygen and methane concentrations are plotted with the minimum oxygen levels and maximum methane levels recorded from the whole windrow (Figure 5.18).

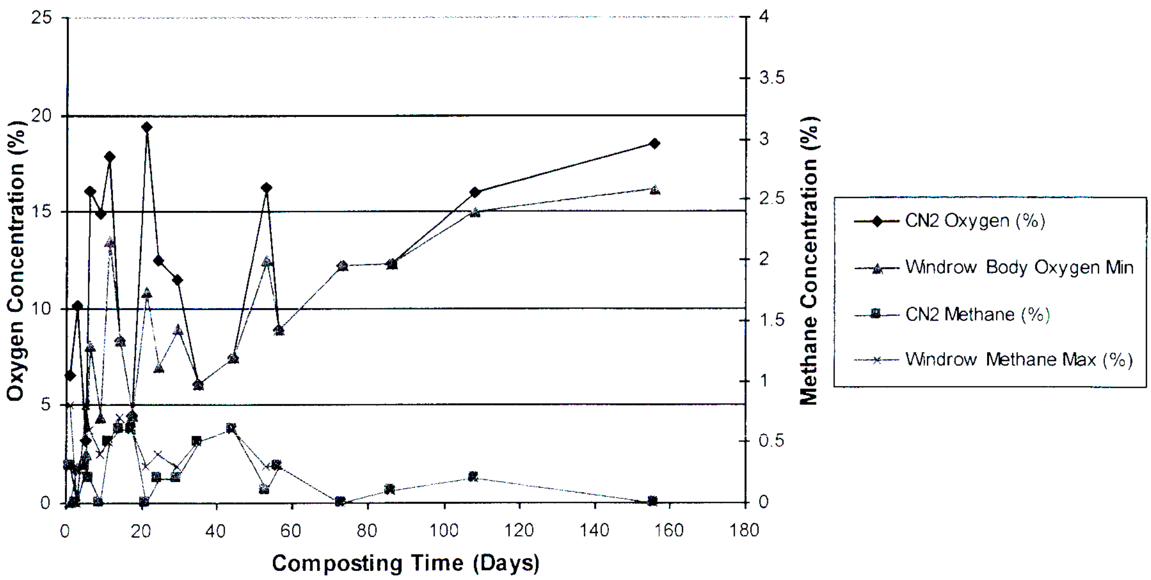

Figure 5.18 Oxygen and Methane Concentrations from Probe CN2

It is evident from Figure 5.18 that low levels in oxygen are closely linked to high levels of methane. This may be due to generation of methane in anaerobic zones, or the detection of methane from the underlying landfilled waste, which is not effectively dispersed in the windrow aeration.

Comparisons

The comparison between the two Durban trials and the Plauen case study (Figure 5.19) shows that Windrow 2 reached higher oxygen concentrations in the shortest period of time. The oxygen concentrations of Windrow 3 were initially higher than those of the Plauen Windrow. However, the concentrations were equivalent for the majority of the composting period.

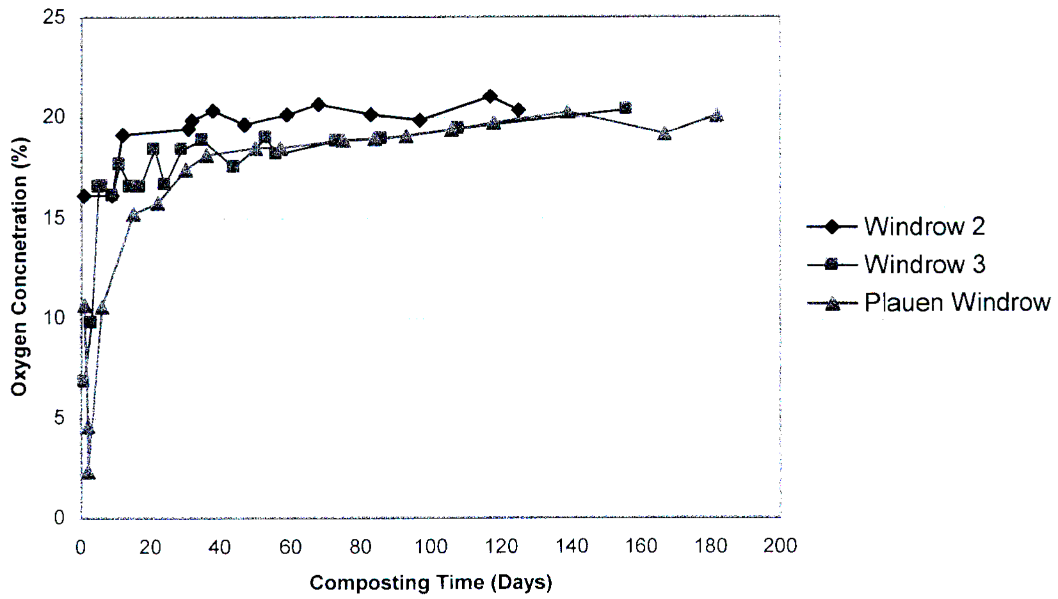

Figure 5.19 Comparative Dome Oxygen Concentrations from the three Windrow Trials

The maintenance of oxygen concentrations close to atmospheric levels in the exhaust gas of Windrow 2 suggests either the highest rate of aeration, or a lowest rate of biological activity. However, when also considering the low temperature difference between the ambient air and dome exhaust gas (Figure 5.10) and the low moisture content of the waste at the end of the trial indicates that a low rate of biological activity occurred in the latter stages of the Windrow 2 composting period.

5.5. AIR FLOW

Windrow 2

The chimney flow velocity for Windrow 2 is presented in Figure 5.20.

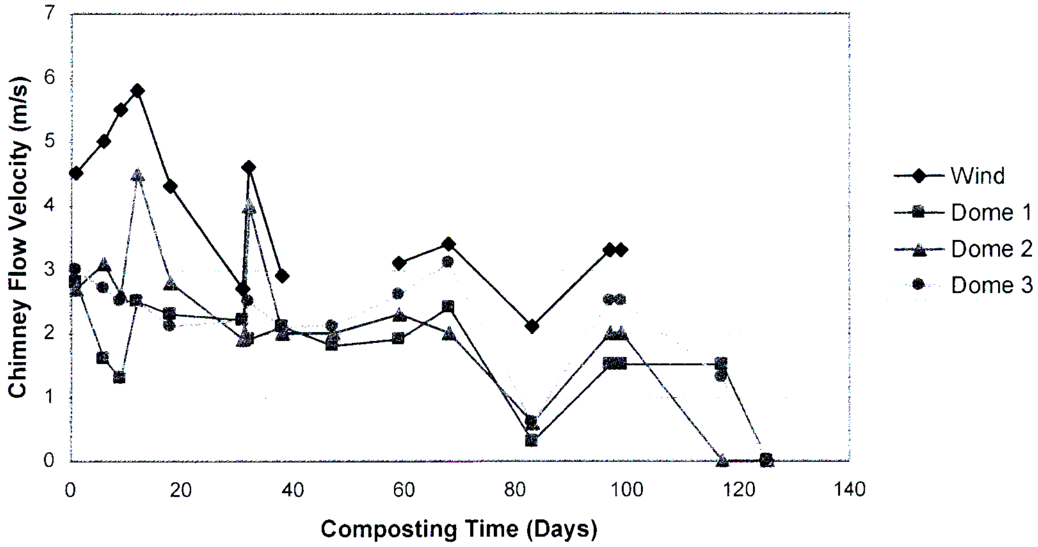

Figure 5.20 Windrow 2 Chimney Flow Velocity

The flow velocity decreases from initial values of 3m/s to end values of approximately 1m/s at the end of the composting period. The wind has a clear effect on the readings, with stronger winds resulting in higher flow velocities.

Windrow 3

The chimney flow velocity for Windrow 3 is presented in Figure 5.21

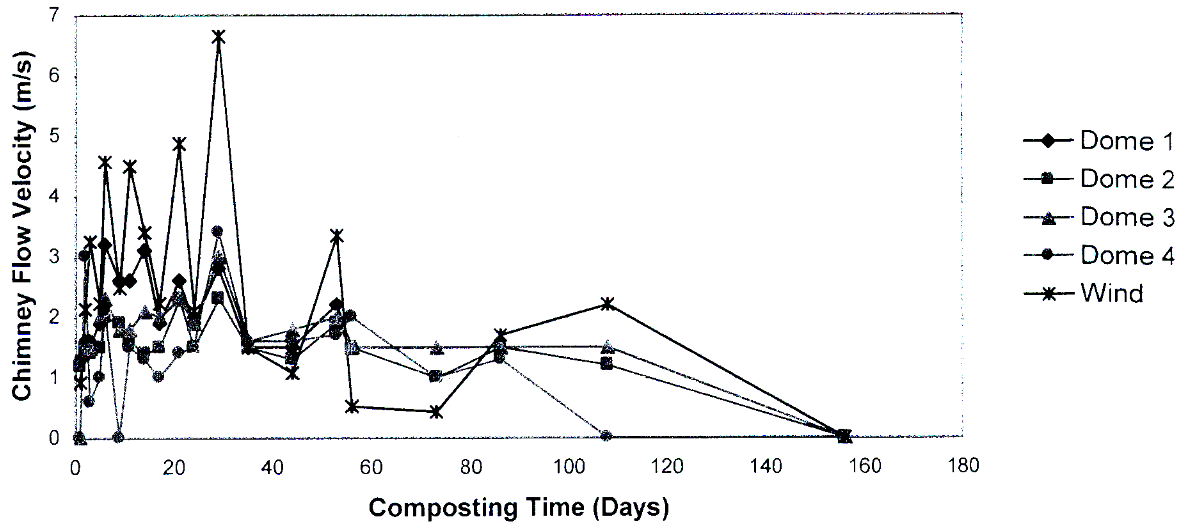

Figure 5.21 Windrow 3 Chimney Flow Velocity

The chimney flow velocity of windrow 3 decreases from the high value of approximately 3m/s to 1m/s over the duration of the compost period. The wind appears to have a significant effect on the chimney flow velocity, as is illustrated by the highest flow of 3.4m/s being recorded at the time of the highest measured wind speed.

Comparisons

The flow velocities of Windrow 2, Windrow 3 and the Plauen Windrow are shown in Figure 5.22 for the purpose of comparison.

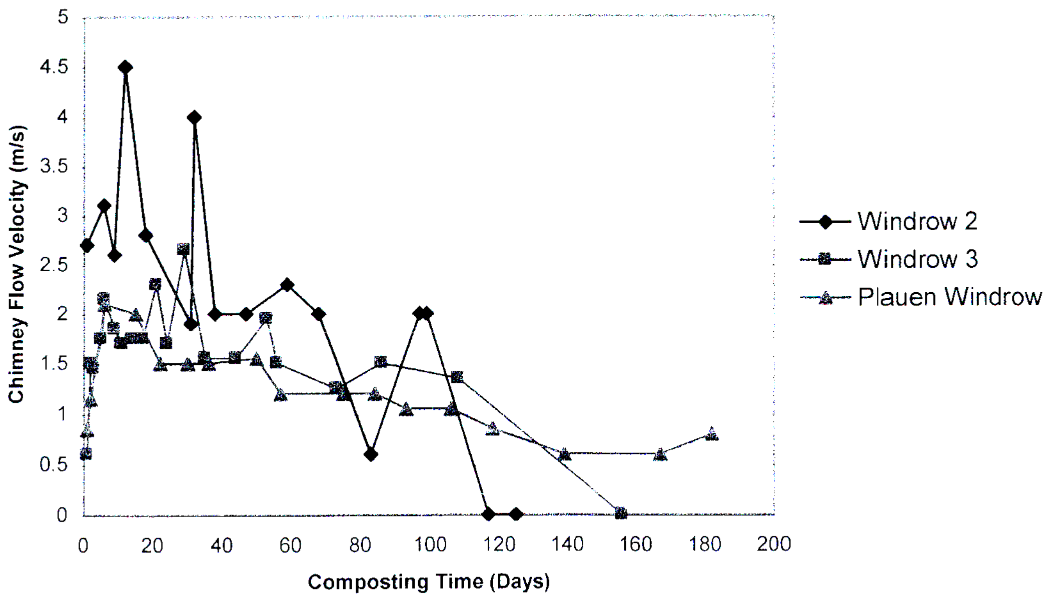

Figure 5.22 Comparisons between Windrow 2, Windrow 3 and the Plauen Windrow Flow Velocity

From Figure 5.21 it is clear that Windrow 2 experiences the highest chimney flow velocities, which corresponds to the highest airflow. The flow velocity of Windrow 3 is close to that of the Plauen Windrow, although the results from Windrow 3 show more fluctuation due to the effect of wind. The reason for Windrow 2 experiencing the highest flow rate despite the lowest temperature difference (Figure 5.10) can only be attributed to the suction effect of the wind passing over the top of the dome chimney. The composting period for Windrow 2 falls within the summer season that experiences higher wind speeds than that of Windrow 3 which fell within the winter season.

5.6. CARBON DIOXIDE PRODUCTION

A means to evaluate the degradation performance of the windrows is by calculating the total carbon dioxide produced during the composting period. This method allows for an evaluation of the entire windrow body, as apposed to discrete sampling of a material as heterogeneous as waste which may have undergone differing degrees of stabilisation, as measured in the poorly ventilated areas. In order to calculate the cumulative carbon dioxide production, the following parameters are required:

- Carbon dioxide concentration
- Intervals between measurements
- Volumetric Flow
- Ambient Temperature
- Exhaust Gas Temperature

The flow of carbon dioxide is the difference between the incoming and outgoing mass of carbon dioxide, plus the carbon dioxide retained within the windrow.

$$M_{CO_2 prod.} = (M_{CO_2 out} - M_{CO_2 in}) + M_{CO_2 ret.} \qquad 5.1$$

It is assumed that there is no appreciable storage of carbon dioxide within the windrow (Paar, 2000).

The incoming carbon dioxide is the atmospheric level of 0,04%. This value falls outside of the measurement range of the gas analyser, which is able to detect to 0,1%. Thus for mathematical simplicity, the incoming carbon dioxide concentration value was assumed 0%. The outgoing carbon dioxide concentration was measured throughout the composting period thus the carbon dioxide production (5.1) simplifies to:

$$M_{CO_2prod.} = M_{CO_2out} \qquad 5.2$$

The mass of carbon dioxide produced is calculated from the following equation:

$$M_{CO_2prod.} = (sV_{CO_2prod.}/Mv) * M_{CO_2} \qquad 5.3$$

where:

$sV_{CO_2prod.}$ is the volume of CO_2 produced at STP (273K, 100kPa)

Mv is the volume of 1 mole of gas at STP : 22,4dm^3

M_{CO_2} is the molar mass of CO_2 : 44g

$sV_{CO_2prod.}$ is calculated by applying Boyle's Law:

$$sV_{CO_2prod.} = \frac{V_{CO_2prod.} * p_m * 273K}{T_m * 100kPa} \qquad 5.4$$

where:

$V_{CO_2prod.}$ is the measured volume of CO_2 produced

p_m is the ambient pressure measured

T_m is the temperature of the exhaust gas

V_{CO_2prod} is calculated by multiplying the measured carbon dioxide concentration [CO_2] to the total volume of gas released from the windrow:

$$V_{CO_2prod} = [CO_2]*V_{tot} \qquad 5.5$$

V_{tot}, is calculated as follows:

$$V_{tot} = 0.85 * v * \frac{\pi}{4} * d^2 * t \qquad 5.6$$

0.85 is the coefficeint for turbulent flow, as determined by Paar (2000).

v is the measured chimney flow velocity

d is the chimney diameter = 110mm

t is time.

As the measurements are not continuous, the calculation is based on the interval between measurements. This requires averaging the measured values [CO_2], T_d and v. The parameters are calculated as follows:

$$[CO_2]av = \frac{[CO_2]_t + [CO_2]_{t+\Delta t}}{2} \quad 5.7$$

$$T_d av = \frac{(T_d)_t + (T_d)_{t+\Delta t}}{2}$$

$$v_d av = \frac{(v_d)_t + (v_d)_{t+\Delta t}}{2} \quad 5.8$$

where the subscripts t and t+Δt denote the values for consecutive measurement intervals

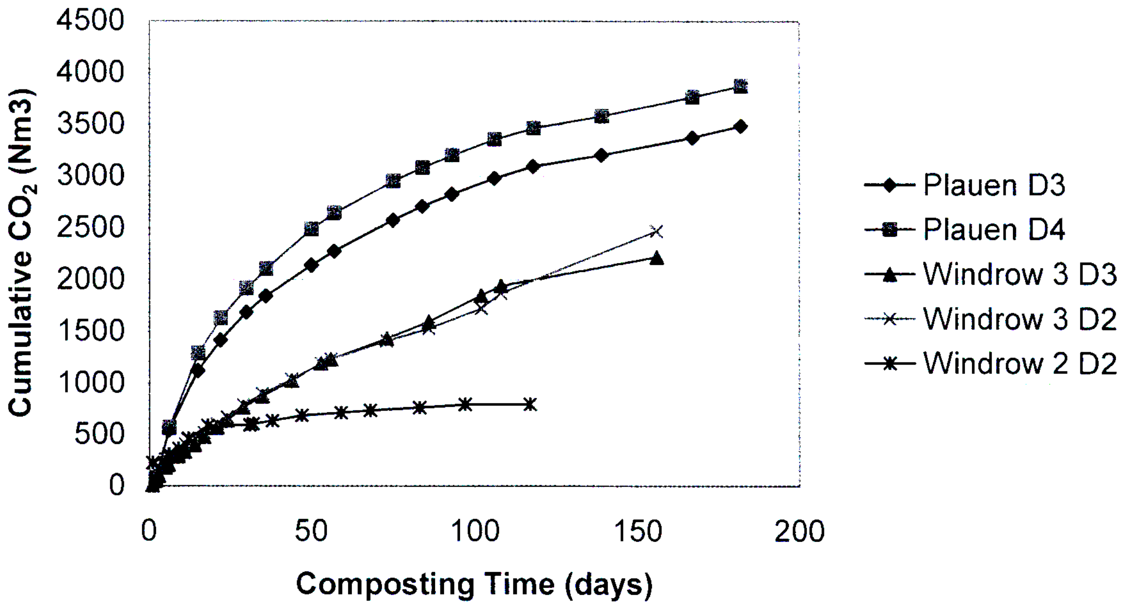

Figure 5.23 Cumulative Carbon Dioxide Production

From Figure 5.23, it is clear that the carbon dioxide production from Windrow 2 slowed dramatically after 18 days. The final carbon dioxide production is comparable to that of Windrow 3 after 35 days.

The difference in ultimate carbon dioxide production between Windrow 3 and the Plauen DAT trials reflects the difference in material input between the two tests. In Plauen a larger ratio of 2 parts MSW to 1 SM was used in the construction of the windrows, as opposed to the 1:1 ratio used for the Durban windrows. Additionally, the Plauen windrows were larger, and thus contained more material per dome. The higher degree of readily degradable organics for the Plauen windrows is evident in the rapid carbon dioxide production in the earlier stages, while the presence of more slowly degradable fractions, such as the large volume of pine bark, is evident in Windrow 3, where the rate of carbon dioxide production is still steady at the end of testing. From the results it is evident that the degradation processes in Windrow 3 were still relatively active. This is substantiated by the higher temperatures recorded for Windrow 3 (Figure 5.9).

Estimated Organic Conversion

Based on the theoretical maximum carbon dioxide production of $4930m^3$, the ultimate organic degradation in Windrow 2 after 20 weeks, $800m^3$ CO_{2prod}, indicates a biological degradation of less than 16%. For Windrow 3, 8 weeks of composting results in a degradation of 24% (approximately $1200m^3$ CO_{2prod}). After 22 weeks ($2300m^3$ CO_{2prod}) 47% of the theoretical biodegradable content has been removed. From Chapter 4, approximately 50% of the orgCb is putricible and the remaining more slowly degradable (wood, pine bark, mixed paper).

Due to the rapid decomposition of putricible materials, the use of a larger ratio of MSW to structural material, as in the Cottbus/Plauen DAT operation, should result in a better overall organic carbon degradation efficiency in the equivalent timeframes.

5.7. WINDROW ODOURS

Although no quantitative measurements were conducted, landfill operators and a sample of people from the community living nearby Bisasar Road Landfill Site were asked to comment on the odour from the windrows. The initial intensive composting stage, characterised by a column of steam from the chimney produced an odour that, although distinctive, was never classified as unpleasant. This odour grew less distinctive until the windrows became effectively odourless. During the dismantling of the windrows, the odour of the treated product was also noted. The material recovered from Windrow 2 was very dry and dusty and produced a slight earthy smell at a close proximity. The material recovered from Windrow 3 at 8 weeks was, moist, hot (approximately 50°C) and still steaming. A more distinctive earthy odour was present, although this too was not offensive. A similar situation was observed when the remaining portion of the windrow was dismantled after 22 weeks, although to a lesser degree. Some areas of the windrow were still hot enough to give off steam, and the material, slightly moist in appearance exhibited the same earthy smell, although only detectable at a close proximity.

5.8. SUMMARY

DAT Implementation

The use of standard landfilling equipment proved to be successful in the mechanical preparation and subsequent construction of the DAT windrows. Although specialised machinery would be more efficient in the operations (as available at the Cottbus Landfill), the advantages of being able to utilise the landfill's existing plant are significant. This allows smaller landfill sites, particularly in rural or peri-urban environments, which relay on limited resources, to operate a DAT Windrow treatment operation with the available machinery, hence reducing capital and operational costs.

Durban Windrow Trials

From the analysis of the three Durban Windrows, the following conclusions can me made:

Windrow 1

The moisture content of the windrow is influenced by rainfall, although to varying degrees with a better distribution of moisture in the homogenous bark section of the windrow. Post wetting of the windrow is a questionable undertaking, although more homogeneous windrow material may improve this if additional moisture input is required during the composting process.

Windrow 2

The biological activity of windrow 2 ceased at a premature stage and it can be concluded that the windrow did not achieve the objective of adequate biological treatment. This is evident in the low temperatures of the maturation stage and the low carbon dioxide concentrations. The reason for this slowing of biodegradation is a lack of moisture available for the aerobic decomposition. Thus, despite a conservative approach being adopted during the wetting stage, the losses associated with wetting using a water truck are high.

Windrow 3

The windrow remained biologically active for the entire composting period. However some areas of the windrow experienced sustained high temperatures, coupled with low oxygen concentrations. This indicates that there were areas in the windrow where aeration was not optimal. The presence of these areas could be as a result of inadequate mixing in of structural material, localised compaction during the windrow construction or settlements after the windrow has been built. The difference between areas of good and poor aeration predicted by the modelled stream flows (Figure 3.5) were not observed during the investigation.

The windrow was still relatively active at the time of dismantling and a longer composting period is necessary if the material is to be composted to the stage of negligible biological activity.

Overall Comments on DAT

Based on the results from Windrow 3, the DAT proved to be a successful method of providing oxygen to an aerobic biodegradation process. The aeration reduction due the higher ambient temperatures was not apparent. The maximum temperatures reached during the intensive thermophillic stage are effectively equivalent, indicating the limiting temperature at which the thermophiles are able to exist.

The limited volumes of waste suitable for use as structural material may cause difficulties during full-scale implementation. Screening the composted product in order to recover such material is one means of mitigation.

A significant aspect is the occurrence of areas of poor ventilation. The worst-case scenario for this effect is the formation of localised anaerobic zones where the biological treatment would be ineffective. It also implies that there may be areas where aeration rate is far higher than the windrow average and these areas may be prone to desiccation as the high airflow strips the material of moisture.

Recommendations

In general, an optimum average concentration of oxygen was measured in the windrows. Oxygen concentration levels of 15% were recorded in the domes after the first week of composting. Therefore, ignoring localised areas of poor ventilation, the aeration rate is too high for the windrows. The conclusion is that a larger windrow with a higher degree MSW, such as that used in Cottbus, can be used.

Additionally, when considering the consequences of the material drying out prematurely, a mitigatory measure may be to control the windrow airflow after the early oxygen intensive thermophillic stage by restricting the chimney area. This would reduce the net airflow through the windrow and consequently reduce the total volume of water that is lost due to evaporation, particularly in those areas where the ventilation is relatively high. This requires further investigation in order to ascertain what the actual effects may be.

Finally, in order to assess the suitability of the DAT for communal sites that generate small volumes of waste, research should be conducted on windrows of smaller size.

CHAPTER 6

EMISSIONS FROM TREATED WASTE
(Materials and Methods)

6.1 INTRODUCTION

Samples of the waste treated using the DAT windrow system were collected and tested in order to characterize their potential for anaerobic emissions once landfilled. This chapter presents the methodology of the analysis of material treated using the DAT windrow technology so as to gauge the effect of MBT on landfill emissions. The methods of analysis are discussed as well as the description of the parameters analysed. The material study consisted of two phases – phase 1 conducted from 2003-2004 using material from the windrows constructed by the author and described in chapter 5, and phase 2 conducted from 2005-2007 using material from the windrows constructed by Simelane (2007). The testing operation is summarized in Figure 6.1.

6.2 TESTING WASTE MATERIALS – APPARATUS RATIONALE

The intention of this investigation is for meaningful recommendations to be made regarding the implementation of this form of waste treatment in South Africa. Thus an in-depth analysis of the material was conducted in order to ascertain the benefit of waste treatment in terms of landfill emission reduction. The focus of this testing was on the gaseous and liquid emissions from the waste, with particular emphasis on the long term leaching of the waste. The assessment at full scale carries with it a large risk to the landfill operator as the benefit will only be known after many years of operation. Operators would be reluctant to implement waste treatment processes, which carry high additional capital costs, without actual knowledge of the final outcome. Thus a lab scale test must be conducted. The study of the anaerobic degradation processes that occur within the waste body dates back many years, with research by authors such as Farquar et al., (1973), Stegmann (1982) Rohrs, (1998) and others providing a reliable understanding of the subject. The use of small and lab scale vessels for the study of long term emissions from waste has been widely documented (Farquar and Rovers, 1973; Stegmann, 1982; Novella, 1995; Leikam et. al., 1997, Rohrs, 1998; Cossu et. al, 2003 and others). Lysimeter studies provide an insight into the biodegradation processes, and allow for the estimation of the polluting potential of the waste (Leikam et. al., 1997).

However, the limitations of timeframe and scale must be overcome. If one were to consider the scale in terms of mass, the lab-scale apparatus must be representative of the primary material components. Additionally, the liquid flux is a limiting factor due to the relatively low liquid flux in a full scale site. Thus, to scale a lab-sized experiment in accordance to a full-scale site would require a very small liquid flux, with volumes too small for leachate analysis. In terms of time, a study that were to carry the same timeframe as a full scale site would require too much time to be feasible and of any benefit in the short term. Thus the experiments must be scaled down significantly in term of size and time and scaled up in terms of liquid flux with extrapolation used to predict the results at full scale. In order to assess the reliability of the extrapolation, three scales are used to assess the same material – eluate tests, column tests and lysimeter tests. The tests are all separated by at least one order of magnitude in terms of mass and liquid flux. It is hypothesized that comparison between the three different scales will provide an indication of the reliability of extrapolation to the full scale. The three different scales are tested with a theoretical full-scale landfill cell (Leikam et. al, 1999), which the results will be extrapolated to, is presented for comparison.

Full Scale Benchmark

- Mass : 12 000kg/m^2 (20m deep site)
- Liquid Flux : 5 litres per week (L/S = 0.0004 per week)
- Time : 50 years

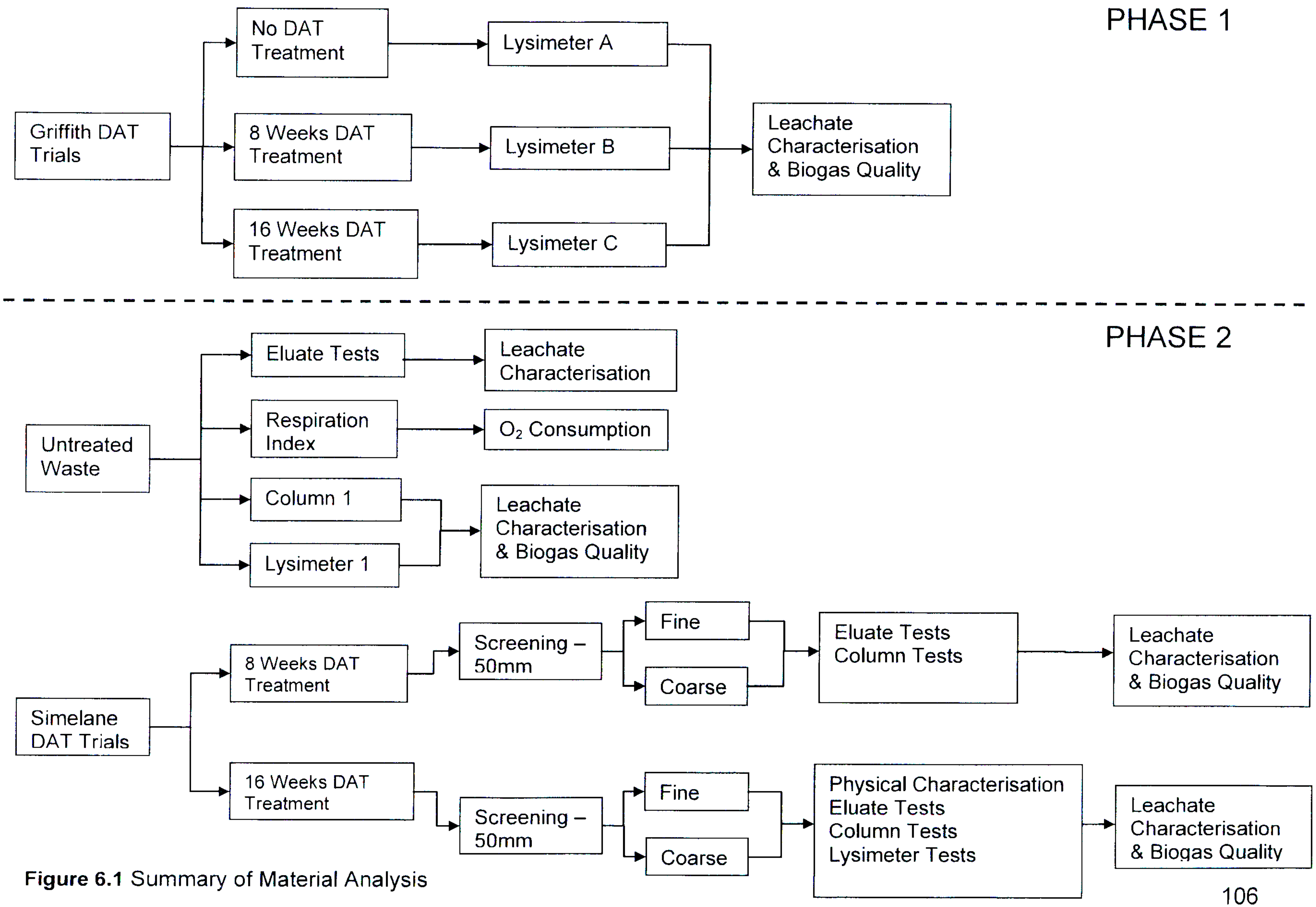

Figure 6.1 Summary of Material Analysis

6.3. PHASE 1

6.3.1 Introduction and Objectives

The landfill emissions from treated waste have been the subject of many studies including those discussed in chapter 2. However, no data was available on the emissions from treated waste in South Africa, a developing country with no source separation of waste. The objective of the testing was to determine the following in the context of the local climate and waste stream:

- The benefit of waste treatment through comparing the emissions from untreated waste to that of treated waste.
- The effect of the duration of the biological treatment step by comparing waste from treatment stages of 8 and 20 weeks.

Three lysimeters were filled with the waste from Windrow 2 and 3 and, a control, with general fresh waste used in the construction of Windrow 3, as outlined in Table 6.1

Table 6.1 Phase 1 Lysimeter Waste Input

	Source	**Composting Period**
Lysimeter A	General waste	None
Lysimeter B	Windrow 2	8 Weeks
Lysimeter C	Windrow 3	20 Weeks

6.3.2 Materials and Methods

The lysimeters consisted of sealed 1000 litre PDE tanks, each with a leachate circulation network, leachate sampling points (Plate 6.1 and 6.2), a gas sampling point, and a probe for temperature measurement (Figure 6.2). The size of the tanks allowed for placement of a waste sample representative of a landfill environment and is comparable to most lysimeter investigations where sizes generally range from 200/ to 1000/ (Farquar and Rovers, 1973; Stegmann, 1982; Novella, 1995; Leikam et. al., 1997, Rohrs, 1998; Cossu et. al, 2003). As the bases of the tanks were flat, a 150mm inverted concrete cone was cast into the base of the lysimeters, in order to ensure the drainage of leachate to a central collection point. A layer of 19mm stone (washed) at the base of the tank and on the surface of the waste allowed for even leachate drainage. The leachate was recirculated each day using pumps coupled to a timer. The overall volume occupied by the waste once placed amounted to approximately 500 litres. PVC piping and fittings were used. A 50mm layer of insulation fabric served to minimize the effect of ambient temperature changes (Plate 6.3). The analysis of the lysimeter involved the sampling of leachate and biogas and temperature measurements. Biogas readings were taken using a Geotechnical Instruments GA94 infrared gas analyser which tested for the volumetric percentage of oxygen, carbon dioxide and methane. The leachate analysis was conducted on a weekly basis: 2 litres of leachate from Sample Point 1, and 0.5 litres from Sample Point 2 (Figure 6.2). The samples were always taken at mid-morning and the analytical work was conducted on site, at the Environmental Engineering Laboratory at UKZN and at certified laboratories. The US Standard Methods for Wastewater Analysis (Clesceri et al, 1989) were used. Parameters tested were pH, conductivity, COD, ammoniacal nitrogen, nitrogen oxides as nitrogen (NO_x-N) and alkalinity, described in 6.5. The overall analysis procedure is summarized in Table 6.2.

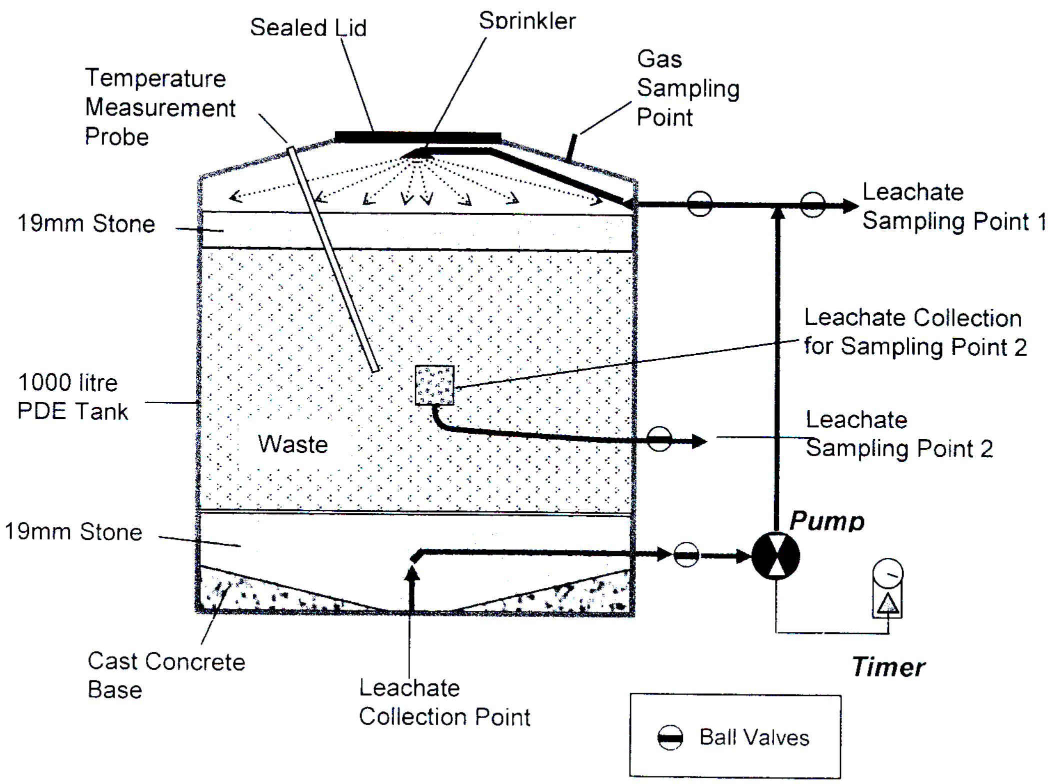

Figure 6.2 Phase 1 Lysimeter Apparatus with leachate recirculation

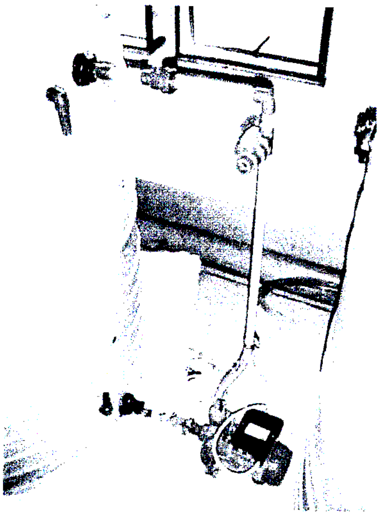

Plate 6.1 Leachate re-circulation network and sample point 1

Plate 6.2 Sample Point 2 collection bucket

Table 6.2 Lysimeter Sampling Details

Sampling Point	**Location**	**Parameters Analysed**	**Frequency**
Leachate 1 (Plate 6.2)	Junction off Circulation pipe.	pH Conductivity	Weekly
Leachate 2 (Plate 6.3)	Collection Point in Centre of Waste Mass	Alkalinity BOD_5* COD Ammoniacal Nitrogen TKN Nitrates and Nitrites Volatile Fatty Acids	Weekly
Biogas	Pipe leading into Lysimeter Headspace	Oxygen Carbon Dioxide Methane	Weekly
Temperature	A sealed pipe projecting into the centre of the waste	Lysimeter Waste Temperature	Weekly

*BOD_5 Analysis was conducted Fortnightly from day 150 onwards.

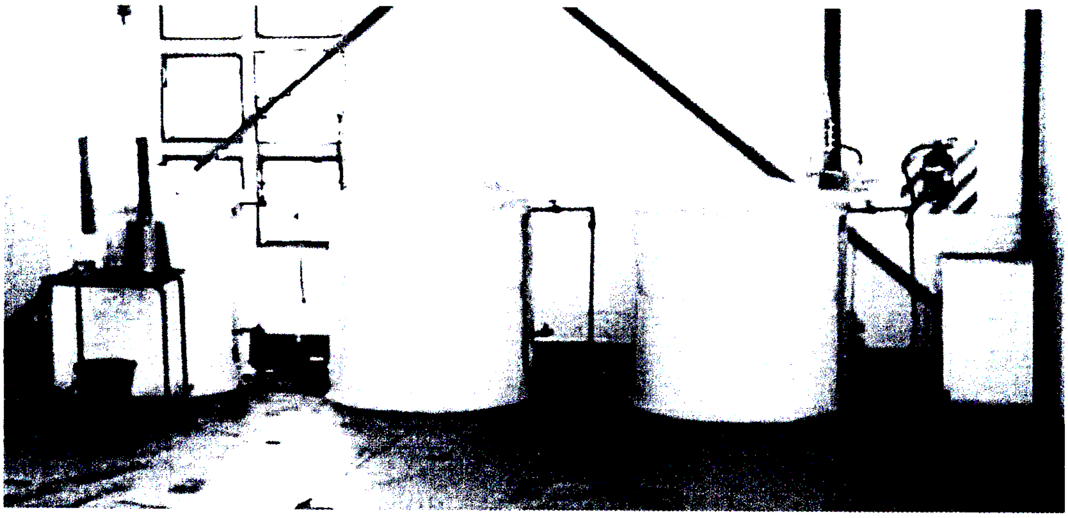

Plate 6.3 Phase 1 Lysimeters

6.3.3 Lysimeters input material

The lysimeters were filled with waste, placed in layers and lightly compacted before being sealed to create anaerobic conditions. The final lysimeter input details are given in Table 6.3.

Table 6.3 Lysimeter Input Parameters

Parameter	Lysimeter A	Lysimeter B	Lysimeter C
Moisture Content (% Dry)	98	60	26
Mass Added (kg)	346	293	193
Dry Mass (kg)	175	183	153
Overall Density (kg/m^3)	852	802	718
Dry Density (kg/m^3)	350	367	307

Water Balance

Closed leachate circulation was used to cycle the leachate through the lysimeter material. This process has been implemented in other lysimeter investigations (Stegmann, 1982; Leikam et. al., 1997; Brinkmann et. al., 1997; Höring et. al., 1999) to enable a more representative leachate sampling, as the circulation homogenizes the leachate. For the effective circulation of the leachate, a surplus of liquid must be available thus the waste must be at field capacity, or the waste would absorb any free leachate. A field capacity of 60% (Blight et al, 1992) was used for the calculation of water required and added to the lysimeters (Table 6.4). The liquid was re-circulated for a period of six days. However, after this period, the waste did not absorb all of the water, and the bottom layers of waste were submerged. The tank was drained of this excess moisture and 20 litres of leachate were poured back for the purpose of re-circulation. The re-circulation was performed daily, with the pumps controlled by an electronic timer. The lysimeter waste field capacity (FC) was calculated using the following water balance:

$$V_{FC} = V_0 + V_{Input} - V_{Removed} \qquad 6.1$$

where :
V_0 is the initial volume of water in the lysimeter waste
V_{Input} is the volume of water added to the lysimeter
$V_{Removed}$ is the volume of water removed from the lysimeter

It must be remembered that the field capacity of a material is dependent on the height of the material above the phreatic surface. As the material inside the lysimeter is at a maximum of approximately 500mm above the phreatic surface (limited by the height of the tank) the field capacity calculated is considered to be higher than that which would be found in a landfill, several meters above the phreatic surface (Bear, 1979). Thus the calculation for field capacity found herein is not an accurate representation of the full height of a landfill site and a more thorough analysis of the material filed capacity should be conducted if accurate results for the field capacity at full scale are required. Such a study was attempted with the assistance of the Technical University of Tshwane using a pressure plate testing apparatus. The results however were too scattered for any conclusions to be drawn. The details of the lysimeter water balance are given in Table 6.4

6.3.4 Moisture Flux

During the leachate sampling, conducted on a weekly basis, 2,5 litres of liquid were extracted for chemical analysis, 2,0 litres from sample point 1 and 0,5 litres from sample point 2. This was replaced by 2,5 litres of distilled water, thus ensuring a constant moisture content in the reactor. The water flux is given in table 6.5

Table 6.4 Lysimeter Waste Field Capacity and Moisture Content

Parameter	Unit	Lysimeter A	Lysimeter B	Lysimeter C
Initial Moisture Content	(%)	49	38	21
Initial Moisture (V_0)	(Litres)	171	110	40
Moisture Added (V_{input})	(Litres)	149	231	210
Moisture Removed ($V_{Removed}$)	(Litres)	69	143	44
Final Tank Moisture (V_{FC})	(Litres)	251	218	206
Final Field Capacity	(%)	59	54	57

Table 6.5 Parameters for Lysimeter Water Flux

Parameter	Unit	Lysimeter A	Lysimeter B	Lysimeter C
Water Flux	l/week	2,5	2,5	2,5
Hydraulic Retention Time	weeks	100	82	87
Water Flow/Dry Matter	l/kg/week	0,0143	0,0163	0,0136
Water Flow/Volume Waste	l/m^3/annum	260	260	260

As discussed above, the removal of leachate for sampling purposes with replacement by distilled water will remove pollutants from the lysimeter at a rate higher than that of precipitation infiltrating into landfill sites. Using an average infiltration rate of 12.5litres/m^3/annum for a 20m deep landfill site (Leikam et. al., 1999) the moisture flux due to sampling in the lysimeters is some 21 times greater in the lysimeters than in a landfill site thus the lysimeter testing is thus indicative of an accelerated leaching process.

The analytical results and processed data are attached in Appendix B, while the analysis of the results is presented in Chapter 7. The three lysimeter tests ran for approximately

300 days from June 2003 to March 2004. In July 2005, Phase 2 was initiated, the methodology of which is presented hereunder.

6.4 PHASE 2

6.4.1 Introduction and Objectives

After assessing the results from the first Phase, it was decided to conduct more detailed tests on the characteristics of the MBT output. An important consideration from the results of both the Windrow testing (Chapter 5) and the lysimeter tests (Chapter 7) is related to the coarse fraction of the treated material. The shortage of coarse material described in Chapter 5 and the presence of high refractory COD loadings encountered during the phase 1 lysimeter tests which was thought to be as a result of the slowly degradable fractions, prompted the interest in assessing the difference between the fine and coarse fractions. Thus the material was subject to a post composting mechanical screening (50mm) (Plate 6.4). Screening the waste is employed in MBT both before and after the biological treatment stage in order to separate the high calorific coarse fraction which is usually incinerated (Soyez et al., 2002; Kuehle-Weidemeier et al, 2003). A screen size of 40-100mm is typically used and the separation of the fractions into the two size distributions (>50mm and <50mm) gives an indication of the physical characteristics as well as the potential differences in post landfill emissions. The primary objectives of phase 2 are as follows:

- To determine, with a greater level of certainty, the benefit of MBT treatment with respect to landfill emissions by comparing the emissions of the MBT output to that of untreated waste
- To determine the effect of screening the waste by comparing the characteristics and emission potential of the fine and coarse materials.
- To determine the effect of the timeframe of the biological treatment step by comparing waste after 8 and 20 weeks of composting.

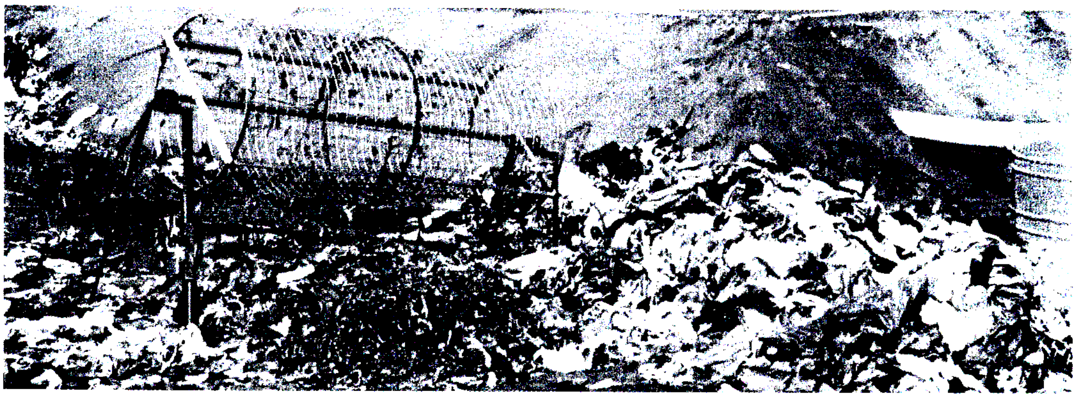

Plate 6.4 The Rotating Drum Screen to separate the fines (<50mm) from the coarse fractions. Note the volume difference between the fine and coarse fractions.

The treated material used in Phase 2 was collected from the DAT windrows constructed by Simelane (2007) while the untreated waste was collected from the transfer station at the Bisasar Road landfill site. The research included the analysis of the following five different material types:

- Untreated Unsorted Waste
- 16 weeks DAT treated waste, <50mm
- 16 weeks DAT treated waste, >50mm
- 8 weeks DAT treated waste, <50mm
- 8 weeks DAT treated waste, >50mm

Numerous tests at three different scales were conducted on the material, as discussed below.

6.4.2 Lab Scale Tests

Physical Characterisation

The proportion of the different material types of each waste stream were analyzed through hand sorting. Additionally, a particle size distribution was conducted on the fine waste using a sieve analysis.

Eluate Tests

The eluate tests were conducted in accordance with the European standards (EN 12457/2 & 4) whereby a sample of size 100g (dry mass) is placed in a vessel and 1000ml of distilled water added to reach a L/S ratio of 1:10. Due to the relatively small size of the sample, the material was shredded to provide a more representative sample. The vessel is then mixed by shaking for 24 hours. Thereafter, the liquid (eluate) is extracted and analysed for the following: pH; conductivity; COD; ammoniacal nitrogen; NOx-N; Total Solids (TS) and Volatile Solids (VS). The dimensions of the tests are as follows:

- Mass : 100g
- Liquid Flux : 1000ml (L/S = 10, once off test)
- Time : 24hrs

6.4.3 Column Tests

Five leaching columns were operated in order to assess the leaching of the material over an extended time frame. The test is similar to the eluate tests in terms of assessing the leaching potential of the material, but the extended time frame allows for the biological solubilization (hydrolysis) to be included in the assessment. The test dimensions are:

- Mass : average 4kg
- Liquid Flux : average 5 litres per week (L/S = 1.2 per week)
- Time : +200 days

Apparatus
The columns consisted of transparent PVC pipe, 160mm in diameter and 1m in length with a volume of 20 litres. Commercial PVC pipe flanges and endplates were used to cap the pipes. A ball valve at the base and at the top of the column allowed for the extraction and replacement of the leachate. Another valve at the top of the column allowed for the connection of the gas measurement burettes which also served for gas analysis. Glass marbles at the base of the column served as an inert drainage layer.

Input Material
The 5 columns contained the following:

- Column 1 – Untreated, unsorted
- Column 2 – 16 Weeks treated, <50mm
- Column 3 – 16 Weeks treated, >50mm
- Column 4 – 8 Weeks treated, <50mm
- Column 5 – 8 Weeks treated, >50mm

Approximately 15 litres of waste sample was weighed and inserted in the column with light compaction. The columns were sealed from the atmosphere to create anaerobic conditions. Plate 6.5 shows the five columns in operation.

Water Balance

The columns were operated in a saturated state, thus ensuring that all the material contained therein was in constant contact with water. Due to the varying densities of the material, the dry mass of material, and thus the liquid to solid ratio, was different for each column. Although this is undesirable when directly comparing the results in the columns to each other, this *modus operandi* was adopted for two primary reasons. Firstly, completely draining the column would provide a more representative leachate sample than if a fraction of the leachate were removed as this may result in plug flow conditions. Secondly, filling the columns to their capacity would be beneficial in terms of increased sample size and consequently a more reliable representation of that material. Table 6.6 summarises the column input parameters.

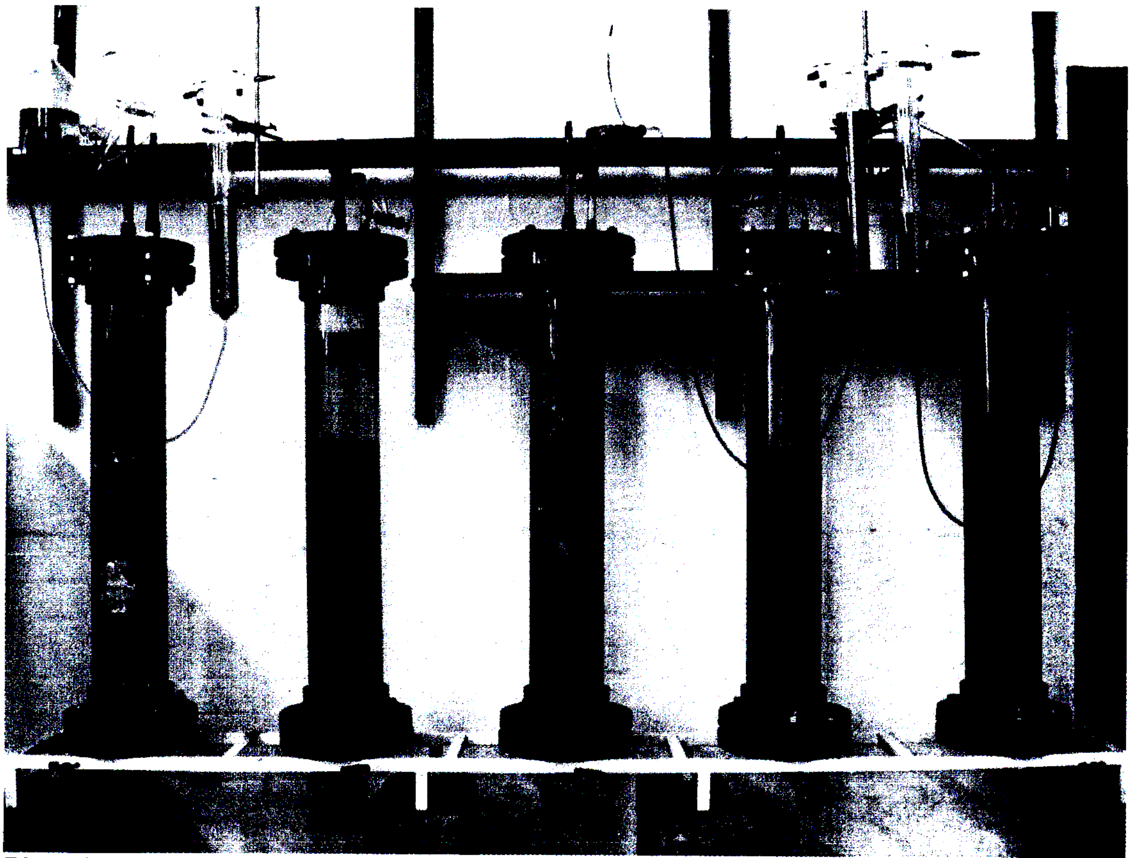

Plate 6.5 From left to right are Columns 1 to 5 – Note the settlement of the fine material in Columns 2 and 4.

Sampling and Analysis
On a weekly basis the biogas quality was tested using an infrared gas analyzer (Geotechnical Instruments – GA 2000) which provided readings of carbon dioxide, methane and oxygen as a percentage of volume. Thereafter the column was completely drained and refilled using distilled water. The leachate was analysed for pH, conductivity, COD, BOD, TS, VS, ammoniacal nitrogen and NO_x-N.

Table 6.6. Column Input Parameters

Parameter	Column 1	Column 2	Column 3	Column 4	Column 5
Waste Type	Untreated, Unsorted	16 weeks <50mm	16 weeks >50mm	8 weeks <50mm	8 weeks <50mm
Dry Mass (kg)	4.95	5.33	1.62	6.19	2.64
Water (litres)	9.57	10.14	12.17	11.83	10.37
Weekly Extraction (litres)	2.50	4.78	7.93	5.66	5.94

6.4.4 Lysimeter Studies

Three lysimeters were operated to test the effects of waste treatment and sorting of waste at a larger scale and in an unsaturated state with open circulation: conditions more similar to those encountered in a full-scale landfill. The dimensions of the lysimeter tests are as follows:

- Mass – average 150kg
- Liquid Flux – average 8 litres per week (L/S = 0.05 per week)
- Time - +430 days

Apparatus
The lysimeters were constructed from PDE water tanks (Figure 6.3). Placement of the tanks on two rings, the outer ring higher than the inner ring, caused the base of the tanks to sag allowing drainage to the centre where a sample tap (PVC ball valve) was located. Washed stone was used as a drainage layer. The material was weighed and placed in the lysimeter with light compaction to a volume of approximately 700 litres (Although for lysimeter 2 the material was highly compressible and thus only 500 litres of material was

placed before the available sample was exhausted. Input water was evenly distributed across the surface of the lysimeter material by means of 3 concentric, perforated rings placed atop a layer of washed stone. The lids of the lysimeters were sealed in order to establish anaerobic conditions and wrapped in thermal fabric insulation. For Lysimeter 1, thermal electric blankets were added during the winter months to prevent possible methanogenic retardation during the colder winter months. A gas vent at the top of the lysimeters allowed for gas measurement and prevented the buildup of high gas pressures and the venting of the gas outside of the building. A ball valve connected to the irrigation rings allowed for water input. A PVC probe allowed for the insertion of a thermocouple for temperature measurement. The lysimeters were filled as follows:

- Lysimeter 1 – Untreated, unsorted
- Lysimeter 2 – 16 Weeks treated, <50mm
- Lysimeter 3 – 16 Weeks treated, >50mm

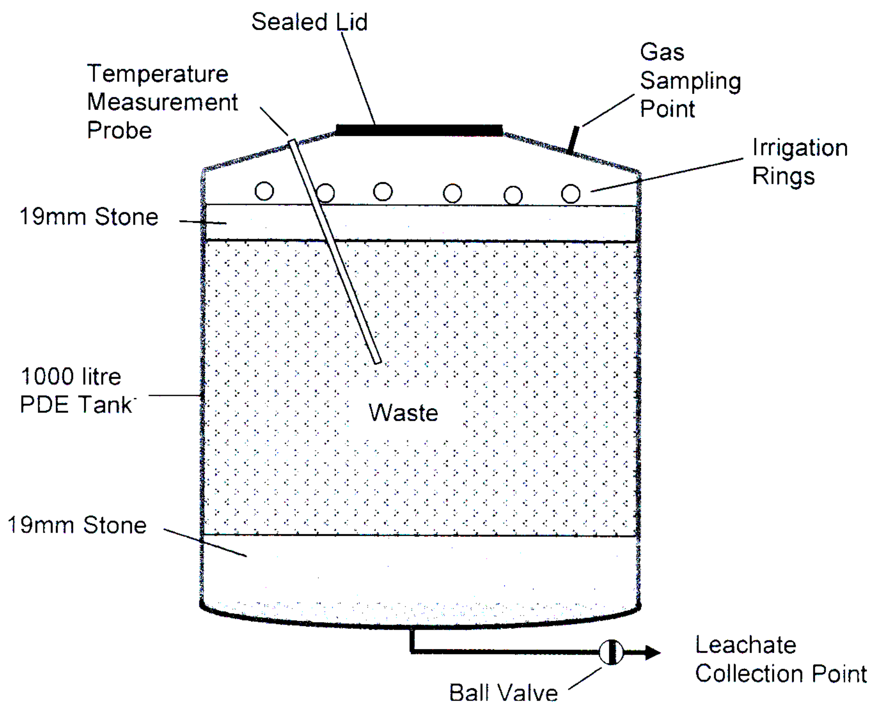

Figure 6.3 Phase 2 Lysimeter Setup. Note the absence of the concrete base.

Water balance

As stated, the lysimeters were operated in an unsaturated state with open leachate circulation. In order to raise the material to field capacity a larger volume of water was added in the initial stages of the tested until a significant proportion of the input water was extracted (80%). After the moisture level of the material was satisfactory, the weekly input of water was adjusted to allow for the attainment of a L/S of 1 after 20 weeks. This corresponds to a liquid flux equivalent to 50 years worth of infiltration into a landfill 20m deep with an infiltration rate of 250litres/m^2/annum. A summary of the lysimeter input parameters is presented in Table 6.7.

Sampling and Analysis

On a weekly basis the biogas quality was tested at mid-morning, thereafter the lysimeters were completely drained and refilled at the fixed rate based on the calculated L/S ratio. The leachate was analysed for pH, conductivity, COD, BOD, TS, VS, ammoniacal nitrogen and NO_x-N.

Table 6.7 Phase 2 Lysimeter input parameters

Parameter	Lysimeter 1	Lysimeter 2	Lysimeter 3
Waste Type	Untreated, Unsorted	16 weeks <50mm	16 weeks >50mm
Dry Mass (kg)	112	283	64
Final Water (litres)	206	191	65
Weekly Extraction (litres)	5.6	14.5	3.5
Weeky Extraction (L/S)	0.05	0.05	0.05

6.5 PARAMETERS ANALYSED

Both Phase 1 and Phase 2 involved the analysis of a range of parameters. The standard method numbers are also given, where applicable

6.5.1 Leachate Parameters

pH - Probe

The pH of a leachate is an indication of the state of biological activity. The initial anaerobic reactions that occur within the lysimeters, the acetogenic and acidogenic phase discussed in (Chapter 2), result in the formation of acetic acid and fatty acids, resulting in acidic pH ranges. As the methanogenic anaerobes become established, these acidic substances are consumed and converted into biogas, causing the pH to rise to neutral and slightly basic ranges. The establishment of a neutral pH thus indicates the existence of a dynamic equilibrium between the acetogienic and methanogenic bacteria.

Conductivity – Probe

The conductivity of a liquid gives an indication of the amount of dissolved ionic compounds; it is also a parameter that may be used to estimate the total dissolved solids (TDS) in the water (Peavy et al, 1985). The specific conductance is the ability of a liquid to conduct electricity, and is a function of its ionic strength. The relation of specific conductance to TDS is not one-to-one, as only the ionized substances contribute to specific conductance and compounds that dissolve without ionizing are not measured.
Despite the lack of distinction in the measured constituents, the conductivity can be used to monitor relative changes in the gross TDS levels of water samples. (Peavy et. al., 1985).

Alkalinity – External Lab

Alkalinity is defined as the quantity of ions in water that will react to neutralize hydrogen ions, and is thus a measure of the ability of the water to neutralize acids. Although there are numerous constituents of alkalinity that may be encountered in natural water systems, the bicarbonate (HCO_3^-), carbonate (CO_3^{2-}) and hydroxide (OH^-) ions are the most common. Apart from their mineral origin, these ions may originate from carbon dioxide – a product of microbial degradation of organic material. Alkalinity measurements were only conducted during the first set of lysimeter tests. Alkalinity tests were conducted by an external laboratory.

Chemical Oxygen Demand (COD) – ASTM 5220B

The amount of oxygen that is required to fully oxidize organic compounds is defined as the chemical oxygen demand. This parameter provides an indication of the polluting potential of the liquid, as the compounds measured by the COD test require further stabilization.

Biochemical Oxygen Demand (BOD) – AI 99005

The organic fraction of the leachate is composed of biodegradable and non-biodegradable organics as expressed by the COD. The BOD test measures only the biodegradable fraction. Although the time frame of a BOD test may vary, a convenient form of the test is the BOD_5, where the measurement is conducted over a period of 5 days. This value is then used to estimate the total BOD of the sample. The BOD_5 is usually found to be 70% of the total BOD (Strachan, 1999).

Volatile Fatty Acids – External Lab

Volatile fatty acids (VFA) are a product of the early acidogenic phase of anaerobic degradation, and thus the measurement provides an indication of the state of biological activity within the waste. Measurements of VFA were only conducted during the first set of lysimeter tests and tested by an external laboratory.

Nitrogen

The four forms of nitrogen that are relevant in the analysis of waste stabilization are organic nitrogen (N_{org}), nitrates (NO_3), nitrites (NO_2) and ammoniacal nitrogen (NH_4-N) – SABS 217:1990. Organic nitrogen is bound in organic compounds. During the process of biological degradation, the complex organic matter is broken down into simpler forms through the activity of microorganisms. An example of this is the breakdown of proteins, into amino acids, and then into ammonia. The sum of ammoniacal nitrogen and organic nitrogen is referred to as total Kjeldahl nitrogen. TKN tests on leachate samples were undertaken by an external laboratory, however, problems soon arose as the TKN readings returned were lower than the ammoniacal nitrogen, a physical impossibility. Investigations by the environmental lab in the school of Civil Engineering at UKZN revealed that the correct digestion of the sample, required for the TKN test, was extremely difficult to achieve, thus the validity of TKN results, particularly on treated waste leachate was questioned and the testing of TKN was abandoned. Interestingly, the same phenomenon was reported by Bone et al (2001) and this implies that the current methods used to determine TKN on treated waste leachates should be reviewed.

In the presence of oxygen, ammonia is converted into nitrates and nitrites, a process known as nitrification (Pelczar et. al., 1977). As nitrification is strictly aerobic, the presence of nitrates and nitrites is an indication of aerobic conditions. Ammoniacal nitrogen can be found in the form of ammonia gas (NH_3) or ionic ammonium (NH_4^+), depending on a dynamic equilibrium as expressed in equation 6.7

$$NH_3 + H_2O \leftrightarrow NH_4^+ + OH^- \qquad 6.7$$

The relative abundance of each species is pH dependent, with more acidic pH ranges (lower OH^- concentrations) pushing the equilibrium to the right, and vice versa. This implies that under the conditions encountered in stable anaerobic environment, the ionic ammonium form will be dominant. Note that in this thesis, the results of the NO_x and ammoniacal nitrogen are presented in the form mg/l of nitrogen or mg/l-N. This is in order to compare the molar abundance of the two forms of nitrogen and if comparisons are made to actual mg/l values, then the results should be adjusted.

Total Solids and Volatile Solids - ASTM B2450

The total solids is the residual content after drying at 105°C indicating the total material content present in the leachate. The volatile solids is the material that is burnt off at 550°C and is generally an indication of the organic content of the TS. The ratio of VS to TS gives an indication of the proportion of organic material to the TS content of the liquid.

6.5.2 Biogas Paramaters

The biogas analysis was conducted using an infrared gas analyser on a weekly basis and tested for the volumetric concentration of the following parameters:

Methane and Carbon Dioxide

The bacterial degradation processes produce carbon dioxide (aerobic and anaerobic) and methane gas (strictly anaerobic). The presence of high concentrations of methane gas indicates that the lysimeter is a stable anaerobic environment, and the relative concentrations of methane to carbon dioxide indicate the level of establishment of the methanogenic bacteria.

Oxygen

The presence of oxygen in the lysimeter is undesirable, as this compromises the strictly anaerobic atmosphere presence in the lysimeter that allows for the desired anaerobic degradation to take place.

Balance Gases

Although the biogas may contain many trace constituents, the majority of this gas is assumed to be nitrogen. Nitrogen is not an active reactant in the anaerobic conditions of the tests, however, the concentration of nitrogen is useful in the qualitative assessment of biogas production. In all the tests there is an injection of nitrogen into the columns or lysimeters during the leachate extraction step. A decrease in nitrogen measured as a percentage of gas volume can only be attributed to the generation of biogas within the test vessel. Thus a decrease in nitrogen concentration is indicative of an increase in biogas production and *vice verse*.

6.6. Sensitivity and Error Analysis

6.6.1. Introduction

The viability of the results is dependent on an acceptable level of accuracy in the analysis. The next section covers the accuracy and repeatability of the lab results. Uncertainties in the results may stem from two basic sources : physical limitations in the testing apparatus and gross variations due to sample heterogeneity and human error. These two aspects will be considered separately, and the results combined to assess the integrity and repeatability of the analyses.

6.6.2. Physical Limitations on Testing Apparatus

Each test is performed using some form of apparatus which has a limitation in accuracy. These limitations must be assessed in order to establish the boundary between certainty and uncertainty in the analysis. In order to perform this task, a sensitivity and error analysis based on a first order Taylor series expansion will be used.

Error Analysis Based on The Taylor Series

Suppose that we wish to evaluate *f(x)*, where *x* is subject to a discrepancy or error (ε). It is useful to assess the effect of ε on the value of the function, i.e., we wish to calculate

$$\Delta f(x+\varepsilon) = |f(x) - f(x+\varepsilon)| \qquad 6.8$$

The problem in this calculation is that the exact value x is unknown. However, from the first order Taylor series expansion, we have:

$$f(x) \cong f(x+\varepsilon) + f'(x+\varepsilon)\cdot(x-(x+\varepsilon)) \qquad 6.9$$

Rearranging 6.9,

$$f(x) - f(x+\varepsilon) \cong f'(x+\varepsilon)\cdot\varepsilon \qquad 6.10$$

Thus, the difference the true value f(x) due to the error ε is

$$\Delta f(x+\varepsilon) = |f'(x+\varepsilon)\cdot(x+\varepsilon)| \qquad 6.11$$

$$\Delta f(x+\varepsilon) = \left|\frac{df}{dx}\right|\cdot\varepsilon \qquad 6.12$$

o

For a multivariable case, the calculation makes use of partial differentials.

$$\Delta f(\tilde{x}_1, \tilde{x}_2, \ldots, \tilde{x}_n) \cong \left|\frac{\partial f}{\partial x_1}\right|\Delta\tilde{x}_1 + \left|\frac{\partial f}{\partial x_2}\right|\Delta\tilde{x}_2 + \cdots + \left|\frac{\partial f}{\partial x_n}\right|\Delta\tilde{x}_n \qquad 6.13$$

$\tilde{x}_1, \tilde{x}_2, \ldots, \tilde{x}_n$ are the values of the variables including the error while $\Delta\tilde{x}_1, \Delta\tilde{x}_2, \ldots, \Delta\tilde{x}_n$ are the errors for each specific variable.

In the context of laboratory results, $\tilde{x}_1$ is, for example, the reading on a piece of equipment while $\Delta\tilde{x}_1$ is the level of accuracy for that equipment.

Dividing through by $f(\tilde{x})$, the relative error (Err) of the result is obtained

$$Err = \frac{\Delta f(\tilde{x})}{f(\tilde{x})} = \left|\frac{\partial f}{\partial x_1}\right| \frac{\Delta \tilde{x}_1}{f(\tilde{x})} + \left|\frac{\partial f}{\partial x_2}\right| \frac{\Delta \tilde{x}_2}{f(\tilde{x})} + \cdots + \left|\frac{\partial f}{\partial x_n}\right| \frac{\Delta \tilde{x}_n}{f(\tilde{x})} \qquad 6.14$$

A very useful tool is the Condition Number (CN) which is a measure of the sensitivity of the function f to each parameter x_i.

The condition number is calculated as follows:

$$CNx_i = \frac{\tilde{x}_i f'(\tilde{x}_i)}{f(\tilde{x})} \qquad 6.15$$

Thus the higher the CN of a certain parameter, the more the function will be affected by variations in that parameter. In the case where the function is expressed as one term only, the CN is independent on the values of the variables and can thus be calculated for all values of the variables x_i. In the cases where the value does effect the CN, the CN will be calculated at a range of values. Combining 6.15 with 6.14, the error (Err) is calculated as follows:

$$Err = CNx_1\left(\frac{\Delta x_1}{x_1}\right) + CNx_2\left(\frac{\Delta x_2}{x_2}\right) + \ldots + CNx_n\left(\frac{\Delta x_n}{x_n}\right) \qquad 6.16$$

Table 6.8 shows a summary of the equipment used during the analysis with the accuracy limits of each apparatus – a crucial component in the accuracy of the results.

Table 6.8 Accuracy of Analysis Equipment

Apparatus	Make & Model	Graduation	Accuracy	Tests
Gas analyser	Geotechnical Instruments – GA 2000	0.1%	0-5% - 1% 5-15%- 3%	Gas Quality
Thermocouple	Major Tech – MT 630	0.1°C	-	Lysimeter Temp
pH probe	Orion – 410A	0.01	±.0005	pH
Conductivity Probe	Corning - 473019	0.1μS/cm	± 0.5%	Conductivity
Spectrophotometer	HACH – DR/2000	0.001A	±0.002A	Closed Reflux COD
4-Point Mass Balance	Denver Instrument Co. - AA 200	0.1mg	±0.1mg	TS & VS
30kg Mass Balance	Nagata – LC2-12	2g	-	Liquid Volume
6kg Mass Balance	Mettle – PE 6000	0.1g	±0.05g	Mass of Solids
Digital Burette	Walv - Continuous E	0.04ml per drop	-	NH_4-N & NO_x-N
Pipettes				
25ml	Generic	Na	±0.03	TS & VS
50ml		Na	±0.07	NH_4-N & NO_x-N
10-100μl	Herschmann Laborgerate -Labopette	1μ*l*	1.0%	CR COD
100-1000 μl		5μ*l*	0.6%	CR COD
1000-5000 μl		50μ*l*	0.6%	CR COD
Measuring cylinders				
25ml	Generic	0.5m*l*	±0.38m*l*	BOD 0-4000
100ml		1m*l*	±0.75m*l*	BOD 0-2000; 0-800
250ml		2m*l*	±1.5m*l*	BOD 0-400; 0-200
500ml		5m*l*	±3.75m*l*	BOD 0-80;0-40
BOD Heads	Aqualytic/WTW -OxiTop	Range Dependant	±1%	BOD

Single Variable Tests

The following tests have only one variable and are limited only by the accuracy of the apparatus. The tests, data range and relative accuracy are presented in Table 6.9.

Table 6.9 Accuracy of Single Variable Tests

Test	**Err**
pH	0.01
Conductivity	0.5%
Temperature	0.1°C
Mass: 0-6kg	0.05g
Mass: 6-30kg	2g
Gas Analyzer	3%

Multivariable Tests

The following tests contain more than one variable which contribute to the uncertainty of the result. These tests are TS & VS, ammoniacal nitrogen and NO_X-N, BOD and COD.

TS & VS

Formulae:

$$TS = \frac{M_{Dry} - M_{Crucible}}{Vol_{sample}} \quad (6.17); \qquad VS = \frac{M_{Fired} - M_{Crucilbe}}{Vol_{sample}} \quad (6.18)$$

Values:

TS: 0 – 20g/ℓ

VS: 0 – 15g/ℓ

$M_{crucible}$: approximately 50g

M_{dry}: 50.000 – 50.500g*

M_{fired}: 50.000 – 50.500g*

*Based on $M_{crucible}$ = 50.000g

Sensitivity:

CN_{Mdry}: 1; $CN_{Mcrucilbe}$: 1; CN_{Mfired}: 1; $CN_{vol\text{-}sample}$ 1

Uncertainties:

$$M_{Crucible} = \pm 0.0001g$$
$$M_{Dry} = \pm 0.0001g$$
$$M_{Fired} = \pm 0.0001g$$
$$Vol_{sample} = \pm 0.03ml$$

Error:

< 0.002%

The high precision of the apparatus gives an extremely reliable result. Note that although there is a slight variation in the Err due to changes in M_{dry} or M_{fired}, these changes are negligible due to the dominance of the Vol_{sample} term.

Ammoniacal Nitrogen and NO_x-N

Formulae

$$NH_4 - N; NO_x - N = \frac{Vol_{HCl} \times 14}{Vol_{sample}} \qquad 6.19$$

Values:

Vol_{HCl}: Range 0.04-100ml

Vol_{sample}: 50mℓ

Sensitivity:

CN_{VolHCl}: 1

$CN_{Volsample}$: 1

The result is equally sensitive to variations in both parameters.

Uncertainty:

Vol_{HCl} = 0,04ml (1 drop)

Vol_{sample} = 0,07ml

The use of a pipette ensures a high level of accuracy in the volume measurement. The uncertainties in the titration volumes (Vol_{HCl}) could be more significant when the titration only requires a few drops.

V_T	volume of the bottle (500mℓ)
V_{sample}	volume of the sample (mℓ)
α	Bunsen absorption coefficient (0,03103)
$\Delta p(O_2)$	difference in the partial pressure of oxygen (mbar)

The only the parameters V_{sample} and $\Delta p(O_2)$ are variables and are thus the only ones that have a CN. The OxiTop heads measure pressure to an accuracy of 1% of the range, ie. 40mg/ℓ for 4000gm/ℓ. A final BOD result of 30% of the range value will be used as a conservative estimate. The values for the CN, uncertainty and relative error are presented in table 6.10

Table 6.10 BOD Test Accuracy Parameters

BOD Range mg/ℓ	V_{sample} Mℓ	CN $_{\Delta p(O2)}$	CN$_{Vsample}$	**Uncertainty** $\Delta p(O_2)$ %	V_{sample} %	**Err** %
4000	21,7		1,04		1,8	5,21
2000	56		1,12		1,4	4,90
800	94		1,22		0,8	4,31
400	157	1	1,44	3.3	1,0	4,77
200	244		1,89		0,6	4,47
80	360		3,29		1,0	6,62
40	428		5,80		0,9	8,55

Sensitivity:

The accuracy of the test is most sensitive to errors in the sample volume measurement at the lower end of the BOD range values.

Relative Errors:

A maximum Err of 8,55% occurs at the lower end of the testing range and is fairly reasonable with overall reliability of the test is a reflection of the sensitivity, with the lowest accuracy found at the lower range values..

COD

The construction of a calibration curve for the spectrophotometer used in the colourmetric closed reflux COD analysis used a series of COD standards. Furthermore, the COD tests conducted during the leachate analysis included the testing of at least one set of standards. This allows the evaluation of the accuracy of the test in absolute terms without the need to assess each piece of equipment individually. This assessment also includes the human operation variations.

Formulae

From the colourmetric calibration curve, COD is calculated as follows:

$$COD = \frac{6232 * Abs}{V_{sample}} \qquad 6.21$$

Relative Errors

The data is scattered about the trend COD=6232*abs with a degree of variation.

It is found that an envelope of ±10% variation in the factor contains 92% of the data (108 data points).

6.6.3. Sample Heterogeneity and Human Error

The other side to data accuracy is the result of human error and sample variations. The nature of the sample material, particularly in the case of solid samples, may result in a significant degree of variation in one data set. The limitation of the human lab worker also plays a role and variations are inevitable in areas where a judgment is required. A few such examples are colour change during titrations and measuring liquid levels against graduation lines. Most of the tests were performed in 3 repeats, with occasionally repeats of 2 and 5 performed, depending on laboratory capacity. Due to the large number of data sets, each with a standard deviation (SD), the SD will be calculated as a percentage of the average value and compiled in a histogram for an assessment of the general trend. These charts are presented in figure 6.5. The analysis is performed on the following tests: COD, BOD, TS, VS, ammoniacal nitrogen and NO_X-N.

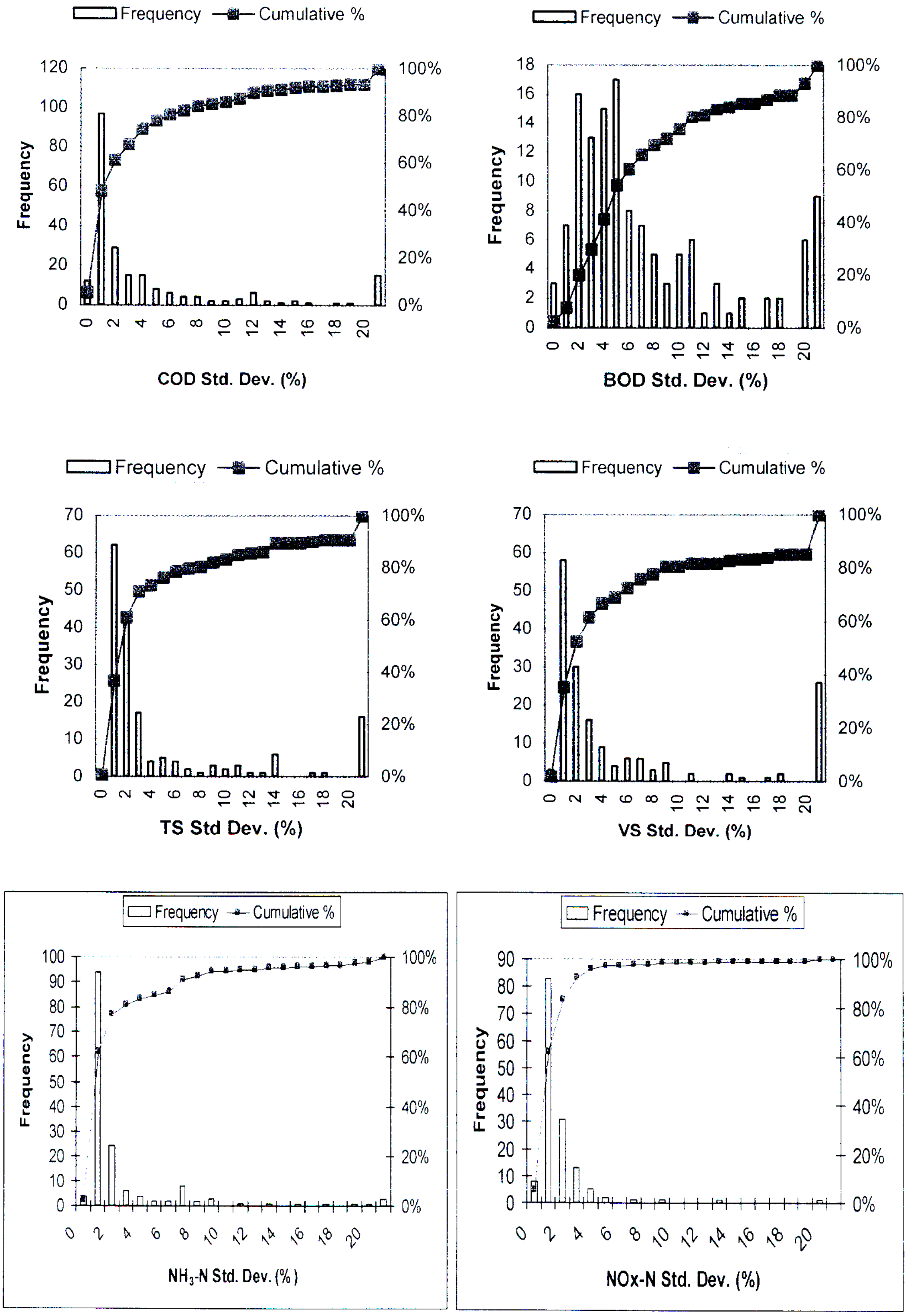

Figure 6.5 Histograms for the standard deviations recorded for various parameter analysis.

Comments

From the histograms it is clear that in general the results are closely grouped about the mean, with the majority of the data showing a SD of less than 5%, with the exception of the BOD results which have a wider scattering. The presence of data with high SD is evident in the TS, VS, COD and BOD results. After closer inspection of these results, it was found that these data were at the lower end of the sample range. Based on the analysis of the histograms, the following final SD's are assumed:

TS:	17%	VS:	20%
NH_4-N:	8%	NO_X-N:	3%
COD:	14%	BOD:	20%

Final Comments

The maximum accuracy of the testing methods used in the laboratory are acceptable with the greatest uncertainty arising from the COD apparatus. The assessment of the SD of the lab results show that all the tests are of acceptable accuracy, with the exception of the BOD test and samples at the lower end of the testing range. Combining the assessment of the Err from the testing and the SD, the ultimate assumed accuracy is presented in Table 6.11.

Table 6.11 Accuracy of Laboratory Tests

Test	Accuracy
Gas Analysis	3%
pH	0.01
Conductivity	0.5%
Mass: 0-6kg	0.5g
Mass: 6-30kg	2g
Temperature	0.1°C
COD	14%
BOD	20%
TS	17%
VS	20%
NH_4-N	7% (for NH_3-N > 7mg/ℓ)
NO_X-N	5% (for NO_X-N > 10mg/ℓ)

The results of all of the anaerobic emission tests are presented in the following chapter.

CHAPTER 7

RESULTS OF THE MATERIAL ANALYSIS

7.1 INTRODUCTION

The results of the analyses described in Chapter 6 are discussed herein. The results include the lab tests, the column tests, the lysimeter tests and the assessment of the leaching of pollutants from the waste which will be used in the development of a leaching or solubilisation model. The results of phase 1 are presented with the finding of this research used in the development of the phase 2 study.

7.2 RESULTS OF PHASE 1

As discussed in Chapter 6, Phase 1 consisted of an anaerobic lysimeter study on three different waste types with material sourced from the Windrows constructed by the author. At the time of analysis, the methodology for the analysis on the solid material had not yet been standardised and thus only the leachate and biogas results are presented. The raw data is available in Appendix B

7.2.1 Lysimeter A

The results from Lysimeter A containing untreated waste from the input into Windrow 3 are presented in Figure 7.1 to 7.3 and discussed.

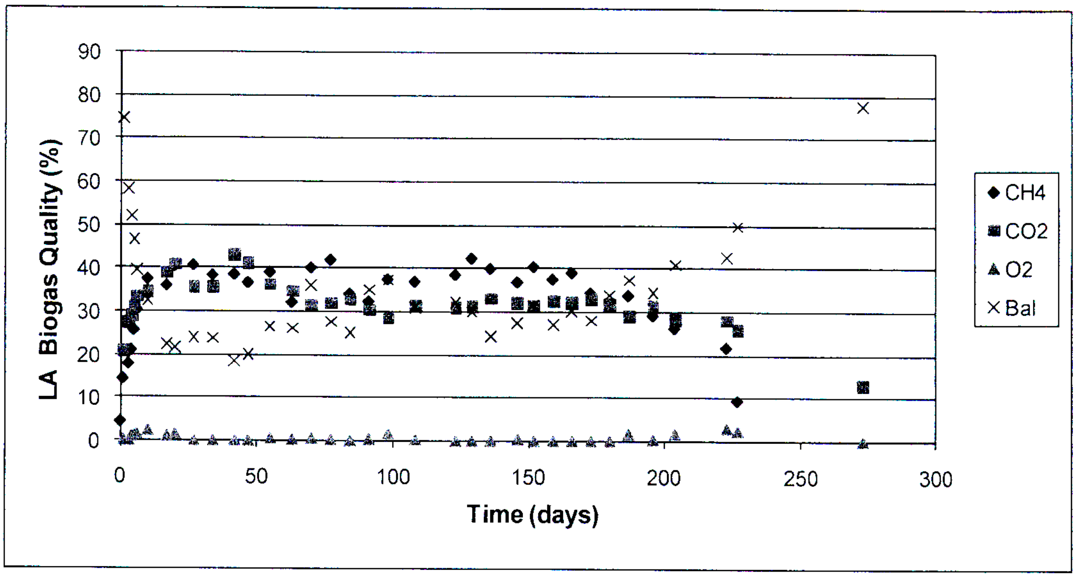

Figure 7.1 Lysimeter A Biogas Quality

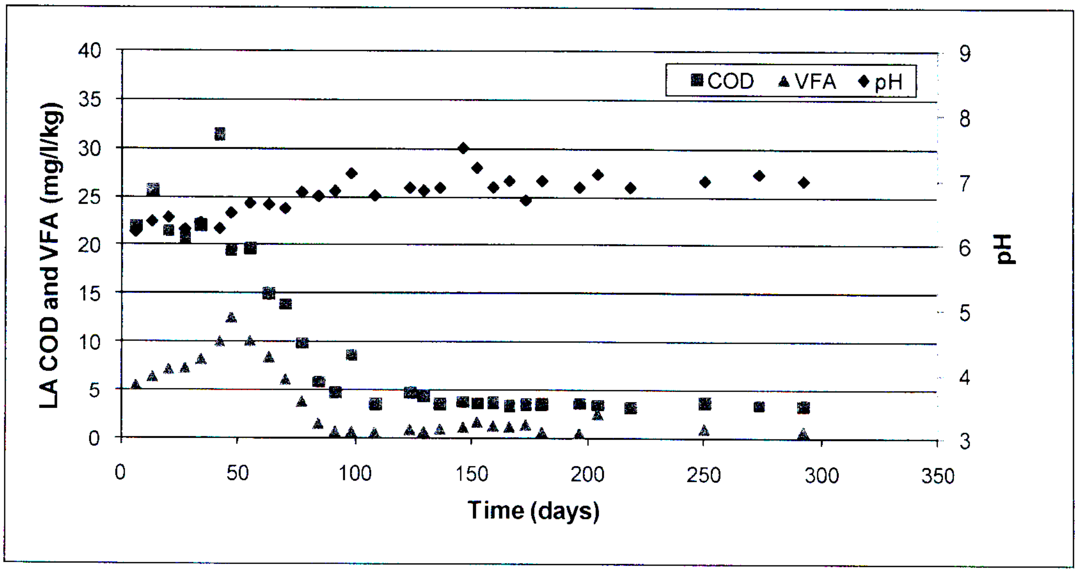

Figure 7.2 Lysimeter A pH, COD and VFA

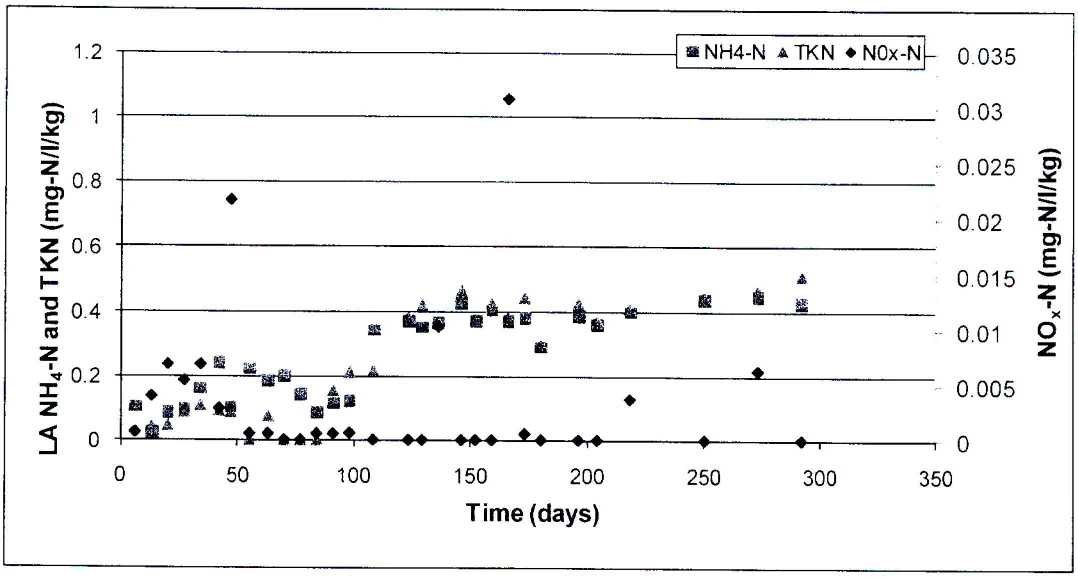

Figure 7.3 Lysimeter A Nitrogen Species

The hydrolysis, acidogenic and acetogenic reactions, characteristic of early anaerobic digestion, are notable at the start of the testing period, with the steady rise in soluble compounds. This is evident from the increase in the leachate conductivity, COD and VFA concentrations, which reach their maximum after 50 days (Figure 7.2). The methanogenic bacteria, reflected in the biogas methane concentration in Figure 7.1, developed rapidly indicating that there was no acidic inhibition of methanogenesis during the early stages of degradation. This is likely to be due to the slightly acidic pH of 6 at the start of the test, as opposed to pH values of 5 that are characteristic of acid phase leachates (Novella, 1995). The subsequent decrease in COD and VFA indicates the activity of the methanogenic bacteria, as the intermediate products of anaerobic digestion are converted into methane, carbon dioxide and water. After 100 days the leachate can be classified as fully methanogenic, with a low COD and a neutral pH. The period after the onset of methanogenisis is generally the stage of maximum biological activity, characterised by a peak in biogas generation. Although there was no measurement of biogas generation volumes, the decrease in biogas concentrations towards the end of the test suggests a slowing of the biogas generation rate. This assumption is supported by the findings of the trial by Novella (1995) where the daily changes in ambient temperatures caused movement of atmospheric air into the lysimeter headspace. At higher biogas generation rates, this would be less evident.

The analysis on the nitrogenous compounds shows a steady increase in the abundance of nitrogen species, predominantly in the form of ammoniacal nitrogen. As stated in Chapter 6, the occurrence of TKN values lower than ammoniacal nitrogen was observed (Figure 7.3). The NO_x-N is detected at very low values.

The high biological activity soon after the commencement of the test indicates the presence of readily degradable organic compounds. However, the pollutant concentrations in the leachate do not reach the same levels as those of other lysimeter investigations into fresh waste (Novella et. al., 1995; Leikam et. al., 1997; Cossu et. al., 2003). The reason for this is the large portion of relatively slowly biodegradable pine bark present in the waste. This may have served as a buffer for the acidic compounds produced during the early anaerobic stages, and lead to the rapid onset of methanogenisis.

7.2.2 Lysimeter B

The results from Lysimeter B containing waste from Windrow 3, which was composted for 8 weeks, are presented in Figure 7.4 to 7.6 and discussed.

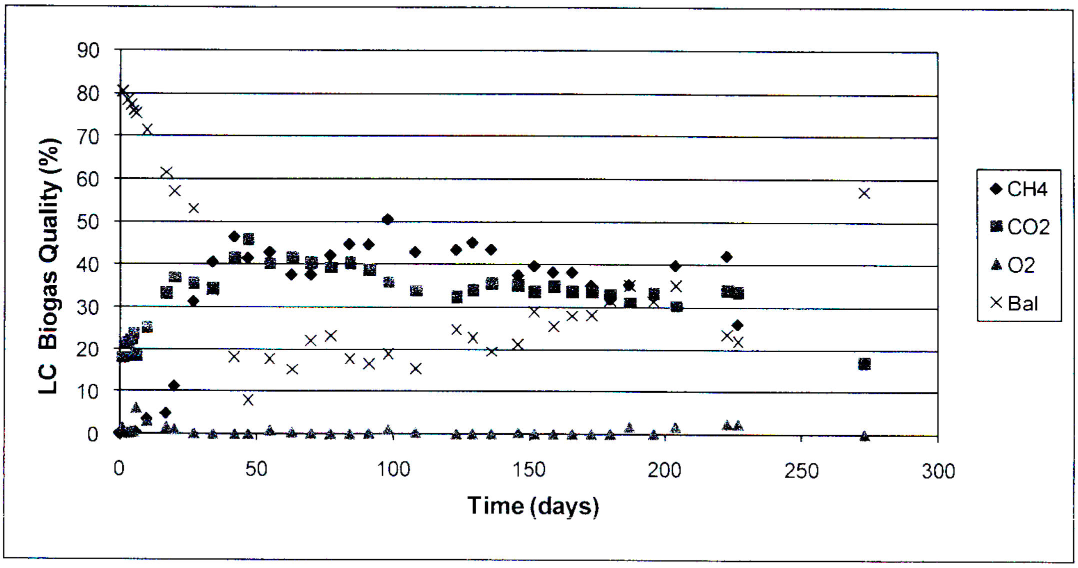

Figure 7.4 Lysimeter B Biogas Quality

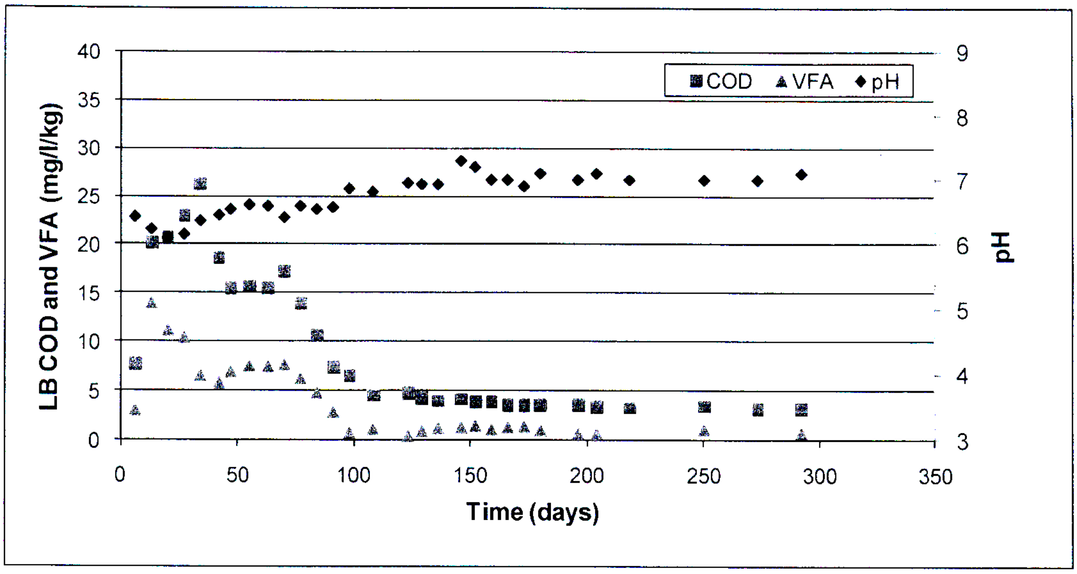

Figure 7.5 Lysimeter B pH, COD and VFA

The typical pattern of anaerobic degradation is followed, with an early peak of dissolved ions (conductivity) and COD and VFA concentrations, which then decrease as the methanogenic bacteria become established (Figure 7.5). There is no acidic inhibition of the methanogenic activity as the lowest pH values are only slightly acidic, although the biological activity is slow at the start of the test indicating lower quantities of readily degradable organic compounds. Methanogenisis is achieved after 100 days, and the drop in biogas concentrations indicates a decrease in biological activity at the end of the test (Figure 7.4). The biogas levels were adversely affected by the breaking of the lysimeter lid seal which would have allowed for the entry of atmospheric oxygen into the lysimeter headspace and caused the dispersion of methane and carbon dioxide. The break in the seal was discovered after approximately 100 days and fixed immediately after which the methane and carbon dioxide levels rose to levels of approximately 30%.

The predominant nitrogen species (Figure 7.6) is ammoniacal nitrogen which rises steadily with time while NO_x-N is detected at very low concentrations. The anomalous measurement of TKN lower than ammoniacal nitrogen was also encountered during these tests. The low levels of NO_x-N indicate that despite the entry of atmospheric oxygen into the lysimeter, aerobic activity was confined to the lysimeter headspace.

A concern with the operation of the lysimeter is whether or not the lack of a proper seal compromised the anaerobic environment. The presence of nitrates and nitrites, recorded on day 42 (4,7mg/l from LB1), day 91 (12mg/l from LB2) and day 108 (3,3mg/l from LB1)

(See Appendix B) indicate that there was aerobic bacterial action, although these values are low and the effect is most likely limited to the headspace area. The most significant impact is on the measured gas concentrations, which would have been affected by the atmospheric air being sucked into the lysimeter while conducting the gas measurements.

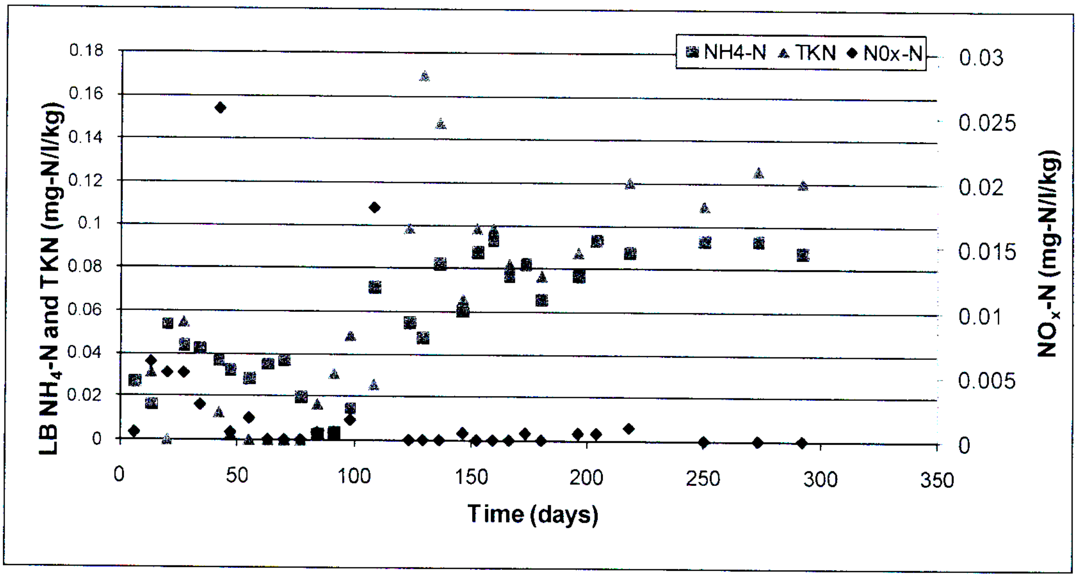

Figure 7.6 Lysimeter B Nitrogen Species

7.2.3 Lysimeter C

The results from Lysimeter C which contained waste from Windrow 2 that had been composted for 20 weeks are presented in Figure 7.7 to 7.9

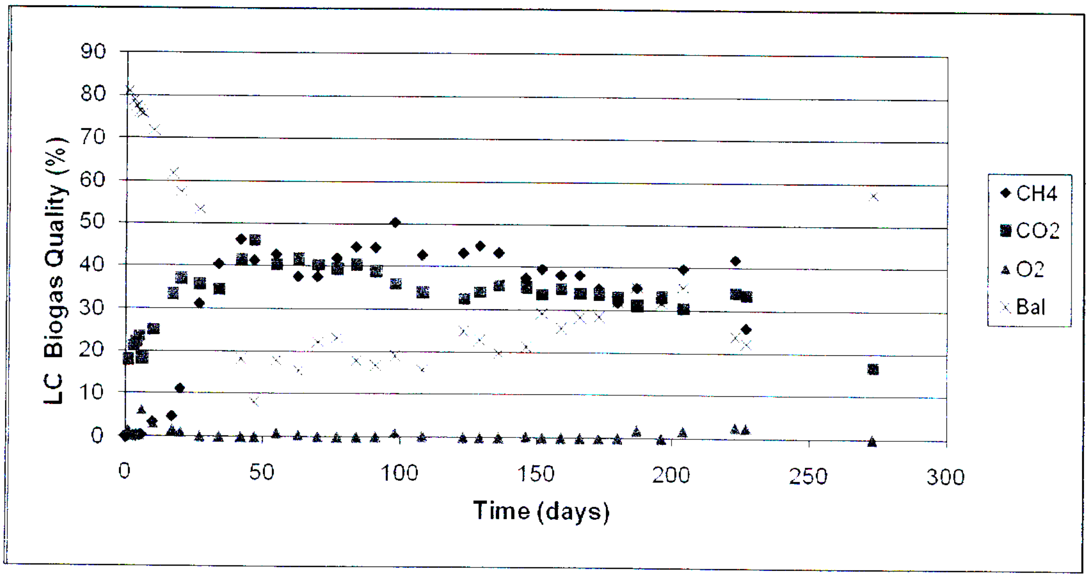

Figure 7.7 Lysimeter C Biogas Quality

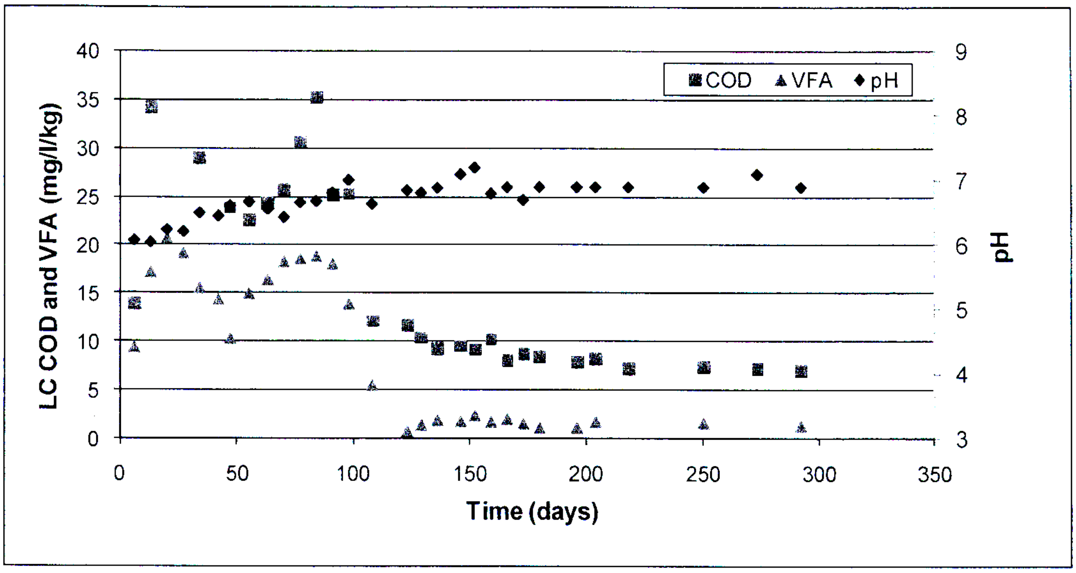

Figure 7.8 Lysimeter C pH, COD and VFA

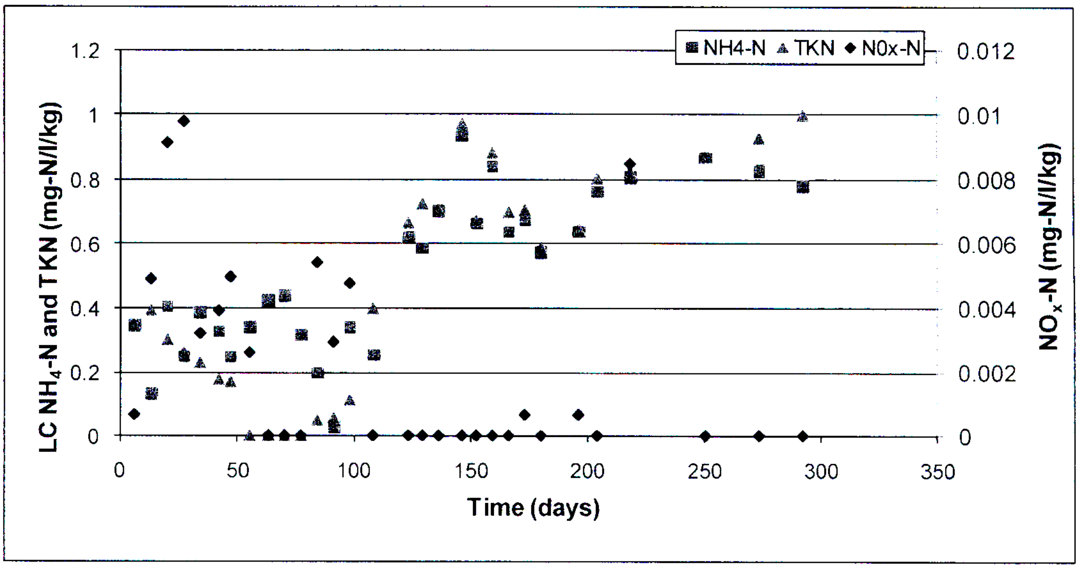

Figure 7.9 Lysimeter C Nitrogen Species

The rapid increase in COD and VFA concentrations reflect activity indicative of early anaerobic microbial development (Figure 7.8). However, the generation of acidic compounds only results in a slight decrease in pH value. The lowest pH (6) is not sufficient to inhibit the methanogenic development and after the residual oxygen had been exhausted, a significant presence of methanogenic activity within the lysimeter is evident after 20 days, (Figure 7.7). The early fluctuation in COD is a phenomenon that can be found in other lysimeter studies (Brinkmann et. al., 1995; Leikam et. al., 1997) and may be a result of the combination of heterorganic microbial activity and inconsistent percolation of the leachate through the lysimeter. The secondary peak in COD and VFA is linked to a slight decrease in methane concentration levels. This suggests that there was a slight inhibition of methanogenic bacteria. It is interesting to note that a secondary COD peak was also encountered on day 50 in the investigation of treated waste by Brinkmann et. al. (1995). The reason for this apparent inhibition in methane levels is not understood, however it may be explained by inconsistent percolation causing a shock load of acid phase compounds to enter into the general leachate flow, causing a slight inhibition of the methanogenic bacteria. The dynamic equilibrium would then be shifted towards the acid phases, compounding the increase in acidic components (High COD and VFA) before the equilibrium once again shifted towards the methanogenic. The onset of full methanogenisis is evident after 120 days, with the decrease in COD and VFA to relatively low stable values and the neutral pH values. The sharp decrease in methane and carbon dioxide concentrations after 250 days indicates a slowing down in the gas production rate which is proportional to the degree of the microbial activity.

As for the other lysimeters, ammoniacal nitrogen is the predominant nitrogen species and rises steadily over time. NOx-N was very low while TKN measurements were at times lower than the ammoniacal nitrogen concentrations (Figure 7.9).

The COD and ammoniacal nitrogen levels are high for a treated waste, confirming that the composting process of Windrow 2 was not effective. However, the lower rate of biological activity at the early stages of the analysis indicates a lower content of readily degradable organic material, which is likely to have been removed during the early intensive thermophilic stage of the composting.

7.2.4 Comparison of Results

The lysimeter investigation provides a basis for comparison of the anaerobic emission characteristics of three different types of waste input – waste that has not been aerobically treated, waste that has been effectively aerobically treated for 8 weeks and waste that has not been effectively aerobically treated for 20 weeks. The results from the testing of the three lysimeters are presented in Table 7.1. The comparison of Phase 1 Lysimeter COD and ammoniacal nitrogen are presented in Figures 7.10 and 7.11 respectively.

Table 7.1 Phase 1 Lysimeter Results Summary

Lysimeter	Unit	A	B	C
Onset of Methanogenisis	days	100	100	120
pH - Lowest	-	6,2	6,0	6,4
pH - Final	-	7,0	6,9	7,0
COD - Initial	mg/l/kg	21.7	7.7	13.7
COD - Peak	mg/l/kg	31.4	26.2	49.0
COD - Final	mg/l/kg	3.4	3.8	8.5
NH_4-N - Initial	mg-N/l/kg	0.10	0.03	0.36
NH_4-N - Final	mg-N/l/kg	0.42	0.09	0.78

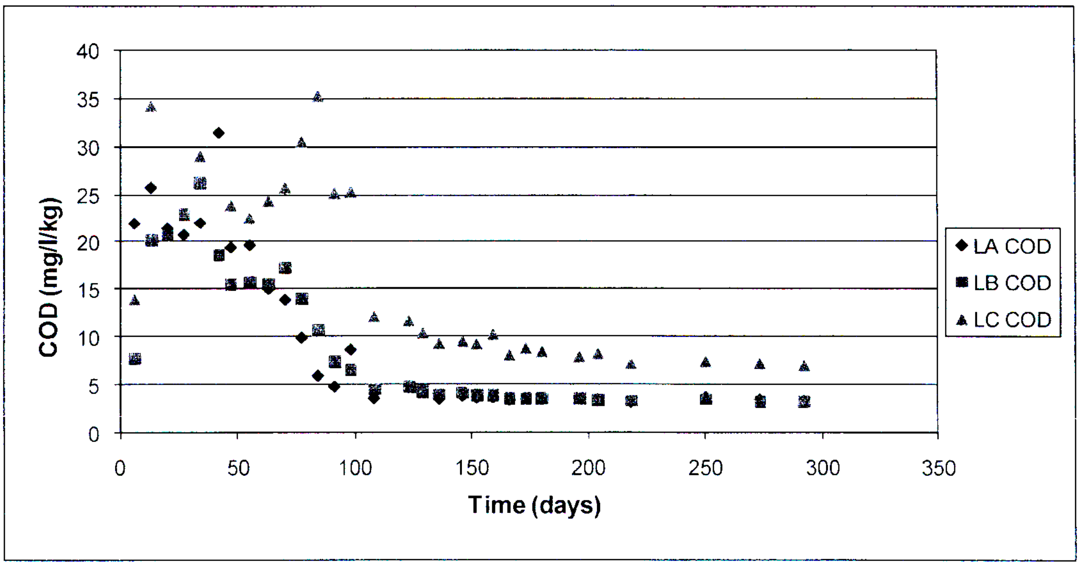

Figure 7.10 Comparison of Lysimeter COD

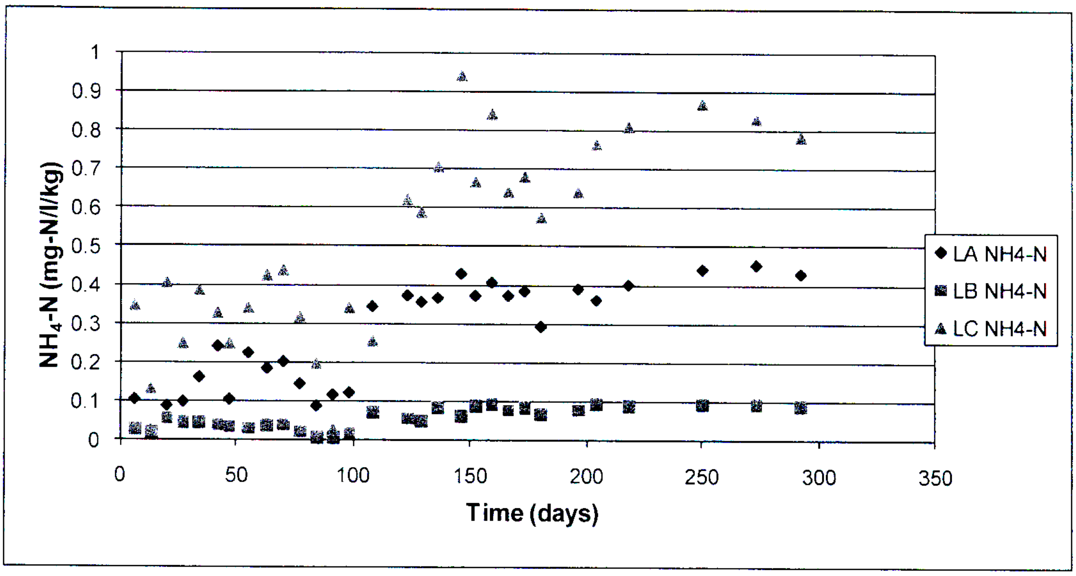

Figure 7.11 Comparision Lysimeter NH_4-N

An observation is the slower increase in methane concentrations in Lysimeters B and C when compared to Lysimeter A. This is explained by the removal of the more readily degradable organic content during the aerobic stage, resulting in a slower removal of oxygen from the lysimeter and longer inhibition of anaerobic bacteria.

The onset of methanogenisis occurs within a similar period of time for the three lysimeters, despite varying organic loading. All three lysimeters showed no sign of acidic inhibition of the development of methanogenic bacteria. The rapid decrease in COD and VFA concentrations demonstrates the effectiveness of the anaerobic microorganisms in the removal of the residual organic compounds. If acidic inhibition can be avoided, then the anaerobic microbial activity can be utilized as an effective biological treatment after landfilling.

Lysimeter A and Lysimeter B

The stable methanogenic leachates from Lysimeter A and Lysimeter B are very similar, particularly the COD and BOD levels. The concentration of hard COD being equal for both lysimeters is as would be expected as the material is from the same source and these compounds would not have been affected during the aerobic treatment stage. The similarity in BOD levels suggests that after the stabilization of the readily degradable organic fractions during the intensive aerobic/anaerobic activity, the remaining slowly degradable fractions are responsible for the BOD levels. The slow degradation of these refractory organic materials, such as paper, bark and wood may persist for many decades (Robinson, 2000). The dependence of the DAT system on a significant proportion of woody waste (used for structural material) may detract from its sustainability. The long-term emission behaviour of these fractions therefore requires further research, as the time frame available for such testing falls beyond the scope of this investigation. If the presence of such fractions does present a problem, the effect may be reduced by screening before landfilling.

A significant benefit of waste treatment is evident in the levels of ammoniacal nitrogen, which are substantially higher for Lysimeter A than for Lysimeter C.

Lysimeter A and Lysimeter C

An unexpected result is found in the comparison between Lysimeter A and Lysimeter C, where the pollutant concentrations are higher for Lysimeter C, which was aerobically treated. This may be as a result of higher putrescible content in the waste of Lysimeter C, or the method used for the storage of the waste used for lysimeter A. The samples were stored in wheelie bins, which were sealed using plumbing tape, at room temperature. It was assumed that the fresh waste would be subject to the acidic inhibition that is characteristic of fresh waste. However, it appears that the pine bark acted as a buffer, resulting in the development of methanogenic bacteria, which would have metabolised the organic material, converting it into methane. The waste was therefore subjected to anaerobic stabilization before being placed in the Lysimeter (approximately 12 weeks), removing some of the organic load.

In the case of Lysimeter C, the waste was subjected to an aerobic treatment equivalent to approximately 30 days before desiccation retarded the microbial activity, effectively preserving the degradable organic compounds in the waste.

Lysimeter B and Lysimeter C

The results reiterate the poor performance of Windrow 2. Thus, despite a windrow period of 20 weeks, the pollutant levels are higher for Lysimeter C, confirming that the effective composting period that was shorter than that of the waste from Windrow 3.

Potential Effect of Concrete Base on Lysimeter Leachate

An aspect that should be considered in the analysis of the lysimeters is the possible effect that the leachate may have had on the unlined concrete base. The concrete may have had a buffering effect if leached by aggressive leachates although inspections of the concrete after the lysimeters were dismantled did not show any erosion of the concrete.

7.2.5 Final Comments on findings from Phase 1 Tests

The major findings from this part of the research are summarized as follows:

- The comparison between the 16week treated waste from Windrow 2 and the 8 week treated waste from Windrow 3 confirmed the suspicions that the performance of Windrow 2 was poor.
- The post methanogenic COD levels of the treated waste were similar to those of the untreated waste, suggesting that the slowly degradable organic fraction, which is not properly treated during the aerobic stage, may be responsible for the long term organic loads in the leachate. This in part motivated the screening of the material for the phase 2 experiments as the coarse fraction is expected to contain the majority of the slowly degradable compounds.
- Anaerobic activity was an effective means of post landfill stabilization with significant COD reductions evident after the onset of methanogenesis. It is assumed that the organic reduction was on readily degradable organics that may have been improperly treated during the aerobic treatment stage.
- The aerobic treatment of waste reduced the levels of ammoniacal nitrogen in the leachate.

The findings and experience gained during phase 1 were used in the design of the phase 2 experiments, the results of which are presented and discussed in the following sections.

Table 7.2 Physical Characterisation

Material Type	Untreated waste		Treated waste		
	DSW Study*	UKZN Study**	**16 Weeks** Un-sieved	**16 Weeks** Fines	**16 Weeks** Coarse
Putrescible	42.5	22	-	-	-
Fines (indistinguishable)	-	-	31	54.5	7.5
Plastic	16.4	31	14.6	2.5	26.7
Paper/Cardboard	19.3	25	13.1	6.3	19.8
Plant/Wood	-	0	9.0	15.4	7.4
Metal	6.9	6	2.8	0.8	4.7
Glass	7.1	7	4.2	0.7	7.7
Other	6.8	-	-	-	-
Fabric	-	4	13.6	1.2	25.9
Stones	-	0	7.7	11.5	3.9
Rubber	-	5	3.8	-	3.5

* Source: DSW in Bowers (2002)
** Source: Marchetti (2007)

7.3 RESULTS OF PHASE 2

As discussed in Chapter 6, the phase 2 analysis was a follow-up of the Phase 1 study into the emissions from treated waste. Phase two investigated the effects of the duration of treatment by comparing untreated waste to waste that has been treated for 8 and 16 weeks, as well as the effect of screening the waste by comparing the fine fractions to the coarse fractions. During this stage, three categories of analysis were conducted:

- The physical and chemical characterisation of the waste
- The analysis of the waste in leaching columns
- Analysis of the emissions from the waste in anaerobic lysimeters

The results of the study are presented hereunder and the raw data is available in Appendix C.

7.3.1 Physical Characterisation

The results from the physical characterization are presented in Table 7.2.

The particle size distribution conducted on the treated fines is presented in Figure 7.12. The ratio of the over-sieved (coarse) to the under-sieved (fines) was approximately 1:2 on a dry weight basis and this is evident when comparing the global samples to the fine and coarse fractions. It was however noted that the coarse material still have contained a portion of fine particles that adhered to its surface and were not removed during the screening.

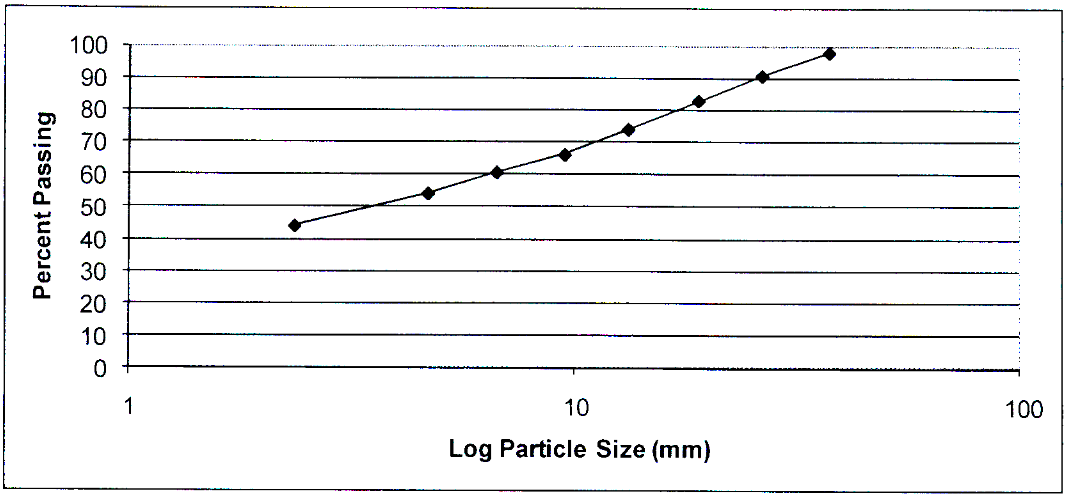

Figure 7.12 Particle Size Distribution of the 16 week treated waste fine fraction

From the DSW physical characterization, it is clear that a large proportion of the untreated material is putrescible, followed to a lesser extent by paper and plastic. The UKZN study indicates a lower proportion of putrescible, however it this figure is likely to be higher due to the cohesion of putrescible matter to the paper and plastic fractions. None of the treated waste samples show any signs of putrescible material with the coarse material mostly comprised of paper and plastic while the fine portion comprised primarily of material too small to distinguish. It is likely that this fine fraction is the humic by-product of the aerobic degradation of the putrescible material. The sieve analysis shows that of the fine waste, 70% has a particle size smaller than 10mm. Note that the treated waste has a higher proportion of plant and wood waste due to the mixing in of structural material during the mechanical preparation stage of the DAT treatment.

7.3.2 Eluate Tests

The results of the 24hr eluate tests presented in Table 7.3 and clearly illustrate the differences in the 5 material types. The untreated waste is acidic with the highest level of dissolved compounds (shown by TS, VS and conductivity) as well as the highest COD.

Table 7.3: Chemical Waste Characterization (±SD)

Parameter	Untreated	8 Weeks		16 Weeks	
		Fine	Coarse	Fine	Coarse
pH	5.3±0.2	7.3±0.1	6.9±0.1	7.3±0.1	7.0±0.1
Conductivity mS/cm	6.2±0.01	1.53±0.16	1.71±0.2	1.41	1.84
COD (mg/l)	7598±131	1489±484	1475±216	3161±304	3640±360
NH_4-N (mg/l)	48.5±30.5	14.63±0.14	10.43±0.14	27.23±0.9	23.3±0.9
NO_x-N (mg/l)	18.1±8.9	8.61±2.1	8.12±1.6	4.93±0.7	5.9±0.7
TS (g/l)	15.62±0.73	2.64±0.79	2.2±0.26	7.31±0.1	5.12±0.1
VS (g/l)	7.13±0.29	1.13±0.41	0.67±0.09	2.62±0.6	2.80±0.2

The nitrogen species tested (both ammoniacal nitrogen and NO_x-N) are found in higher concentrations when compared to the four treated waste samples. pH and conductivity of the treated samples are all similar, however COD, TS, VS and ammoniacal nitrogen are higher in the 16 weeks treated sample, although the NO_x-N is lower. This is an unexpected result as the extended duration should result in lower values of these parameters. This is most likely due to inefficiencies in the DAT process due to desiccation of the 16 week windrow noted by Simelane (2007), while the 8 week sample was taken from a windrow that was irrigated. The accuracy of the results is satisfactory, with acceptable standard deviations recorded for all results, excepting the ammoniacal nitrogen of the untreated waste and the COD of the 8 weeks fines which may have occurred as a result of sample heterogeneity. This must be remembered in the analysis and discussion of the results.

7.4 RESULTS OF THE COLUMN TESTS

The results for each individual column are presented in the following section. Due to the variations in L/S, the different levels of dilution will affect the concentrations of measured parameters. Thus, at this stage, each column is analyzed individually with time as the independent variable. A more direct comparison will be possible when assessing the leaching of the pollutants, presented in chapter 8. A key aspect to observe is the development of methanogenic conditions and any correlation between the various parameters. The results are presented as mg/l per kilogram of total dry solids, where applicable.

7.4.1 Column 1

The results from the analysis of the untreated waste in Column 1 are presented in figures 7.13 to 7.6.

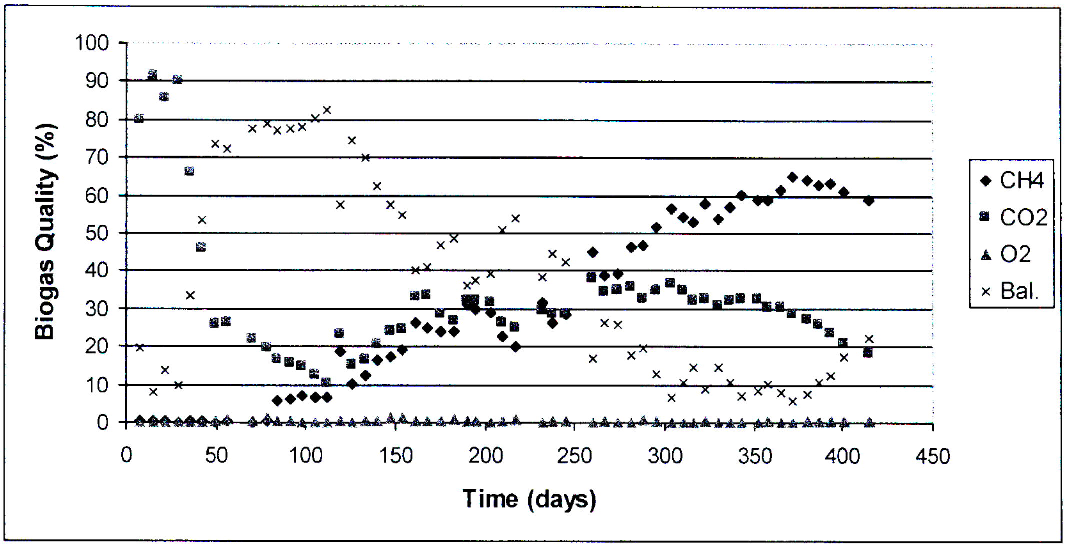

Figure 7.13 Column 1 biogas quality

The biogas quality (Figure 7.13) indicates a short early stage of high-level, non-methanogenic anaerobic activity with high levels of carbon dioxide (90%) and no methane recorded. The concentration of carbon dioxide dropped significantly after this early stage, followed by a slower increase in biogas, with both methane and carbon dioxide levels showing a steady increase from day 100. Methane concentrations continued the steady increase to a maximum level of 60% with carbon dioxide peaking at 35%. The concentrations of biogas suggest that the generation rate increase steadily from day 100 onwards as the methanogenic bacteria became established. The biogas generation appears to peak after 300 days, with the nitrogen levels dropping to around 10% until day 375 where the volume of nitrogen once again begins to rise. The concentration of oxygen remains low (less than 1%) throughout the test.

The characteristic acidic inhibition is evident for the first 80 days with no methane recorded in any significant concentrations. Interestingly, the pH only rises above 6 after 350 days, although significant methane generation commencing with a pH of less than 5 (Figure 7.3). It is generally accepted that a pH lower than 6 inhibits the development of methanogenic bacteria. It must be assumed that within the column exist zones of favorable pH (>6) with the mixing of these basic zones with the acidic areas resulting in an overall acidic pH (<5).

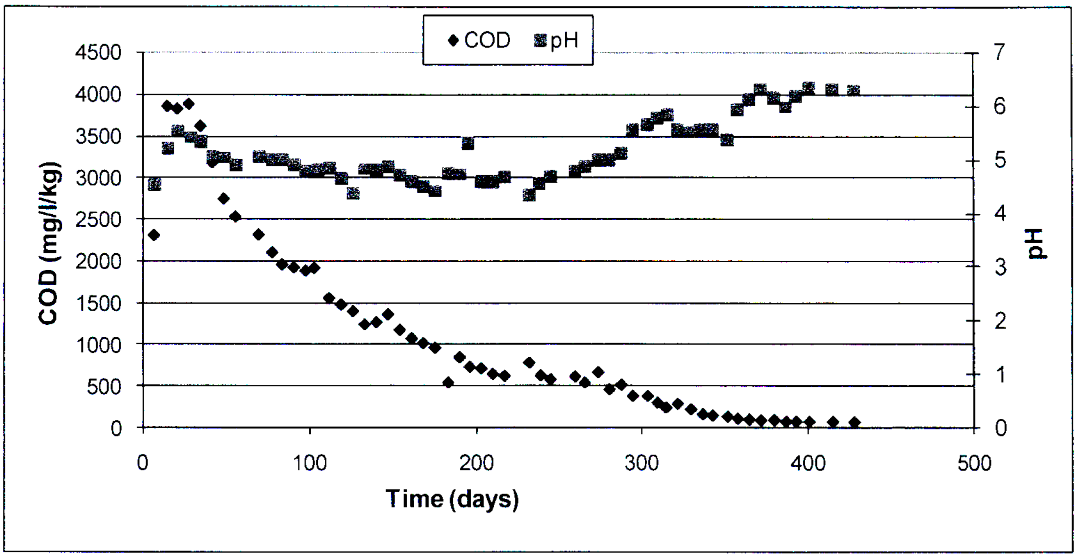

Figure 7.14 Column 1 pH and COD

The COD levels, 4000mg/l/kg at the start of the test, drop steadily throughout the test, with a final recorded value of 60mg/l (Figure 7.14). The shows a discontinuity at approximately day 280 suggesting a change in the biochemical processes and coincides with the rise in pH levels indicating that the column is entering the fully methanogenic stage.

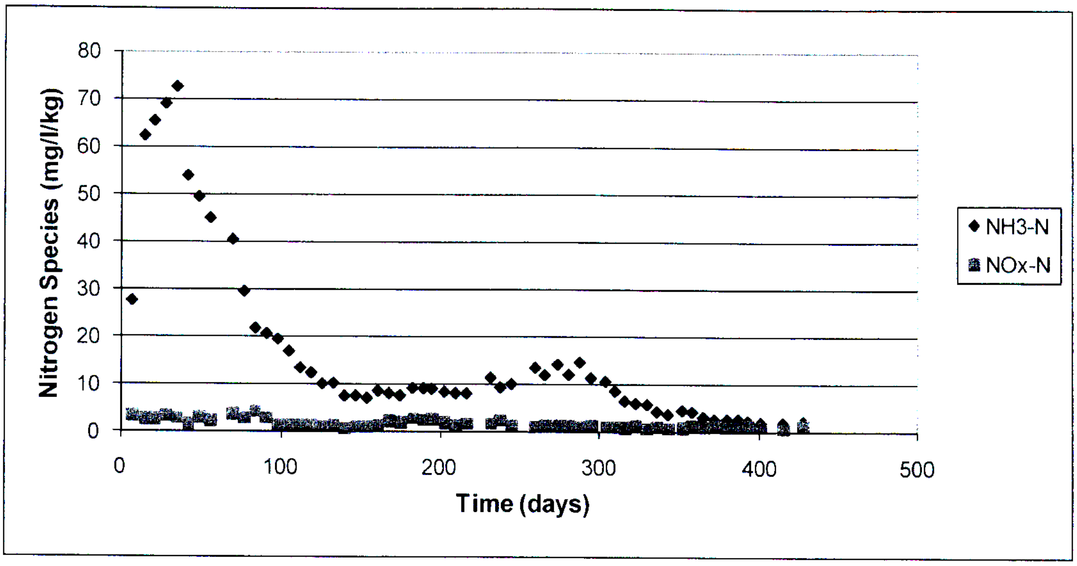

Figure 7.15 Column 1 NH_4-N and NO_x-N

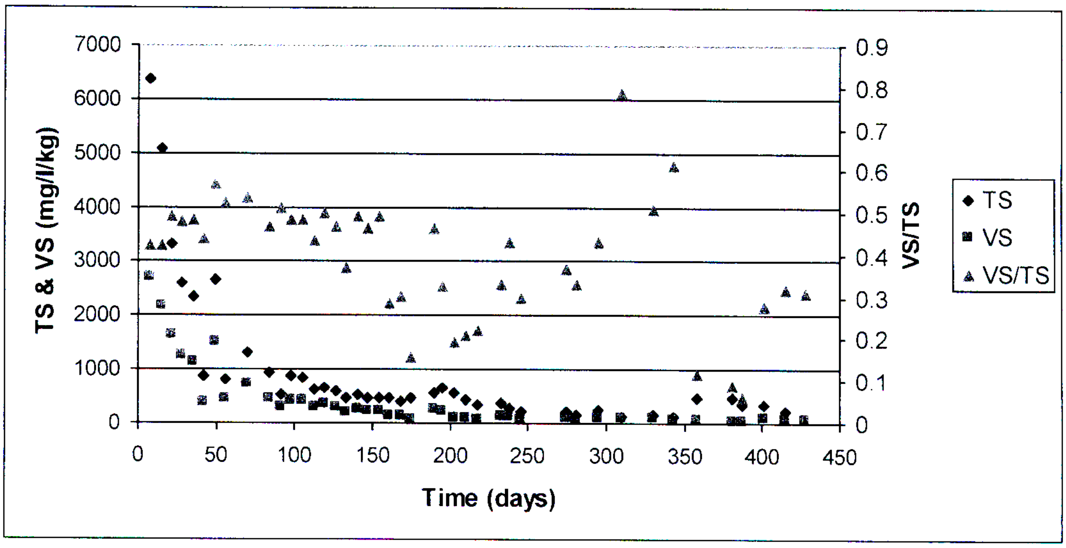

Figure 7.16 Column 1 TS and VS

Of the two nitrogen compounds tested, ammoniacal nitrogen is the predominant species (Figure 7.15), with 70mg/l/kg measured at the start of the test, dropping rapidly to 7mg/l/kg after 150 days before rising slightly for the next 150 days. 280 days into the test the ammoniacal nitrogen levels drop steadily to the low value of 2mg/l/kg measured at the end of the test. The ammoniacal nitrogen levels appear to correlate with the biogas measurements, with the increase rise in values from day 150 corresponding to the start of significant methane generation and may be as a result of enhanced hydrolysis of proteinacious compounds. The second drop in ammoniacal nitrogen appears to correspond to the stage of maximum biogas generation noted from days 300-380. The NO_x-N readings are low throughout the test.

The TS and VS recorded initial values of 6400 and 2700 mg/l/kg respectively (Figure 7.16). These readings drop significantly over the first 100 days of the test to values of 800mg/l/kg (TS) and 400 mg/l/kg (VS). The values decrease over the next 320 days of testing with final values of 180 mg/l/kg measured for TS and 55mg/l/kg VS. The VS/TS relation shows a downward trend, although the result is highly erratic.

7.4.2 Column 2

The results of the 16 week treated fine fraction tested in Column 2 are presented in Figures 7.17 to 7.20 and discussed.

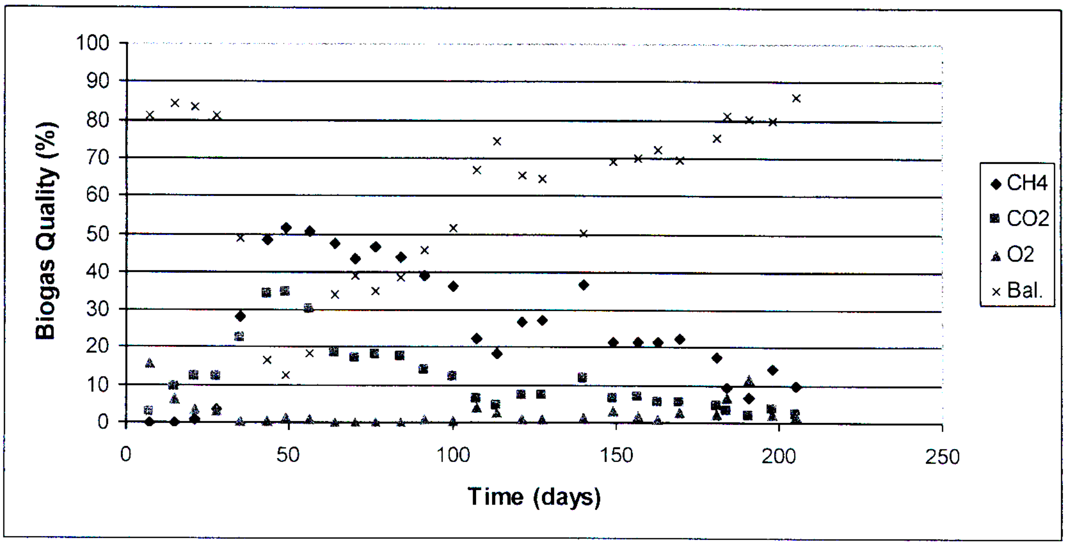

Figure 7.17 Column 2 Biogas Quality

The biogas quality shows high concentrations of methane bacteria at an early stage (Figure 7.7). The maximum values of 52 and 34% for methane and carbon dioxide respectively are reached after 50 days, decreasing steadily thereafter as the biogas generation. This rapid development of methanogenesis indicates an absence of the acidic inhibition stage which is considered to be a characteristic of treated waste

(Robinson 2005). Oxygen is consumed in the early stages of the test and remains negligible throughout the test. The lack of an acid stage is also evident in the pH which remains above the lower limit of 6 and remains within the range of 6-7 throughout the test (Figure 7.18).

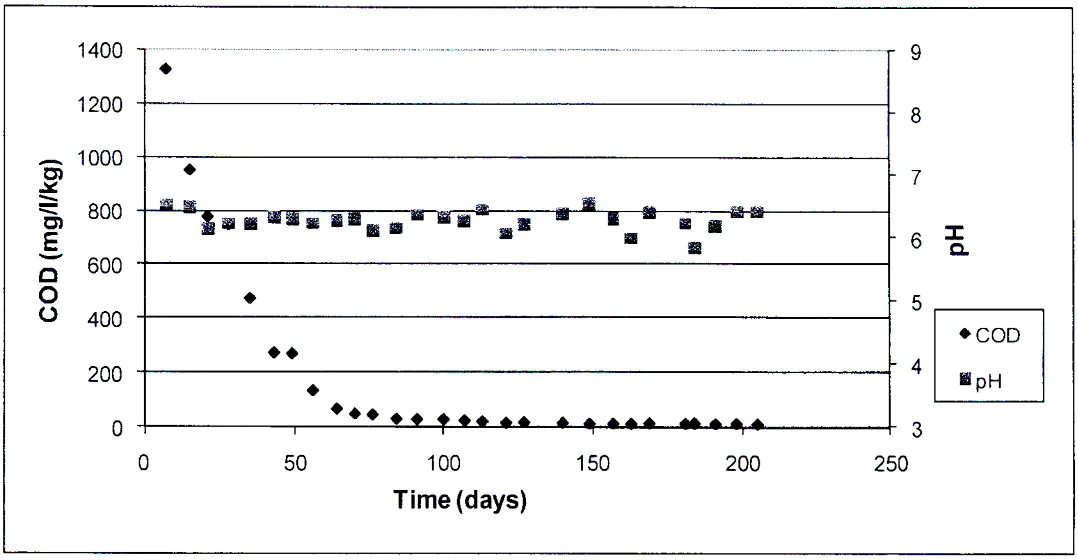

Figure 7.18 Column 2 pH and COD

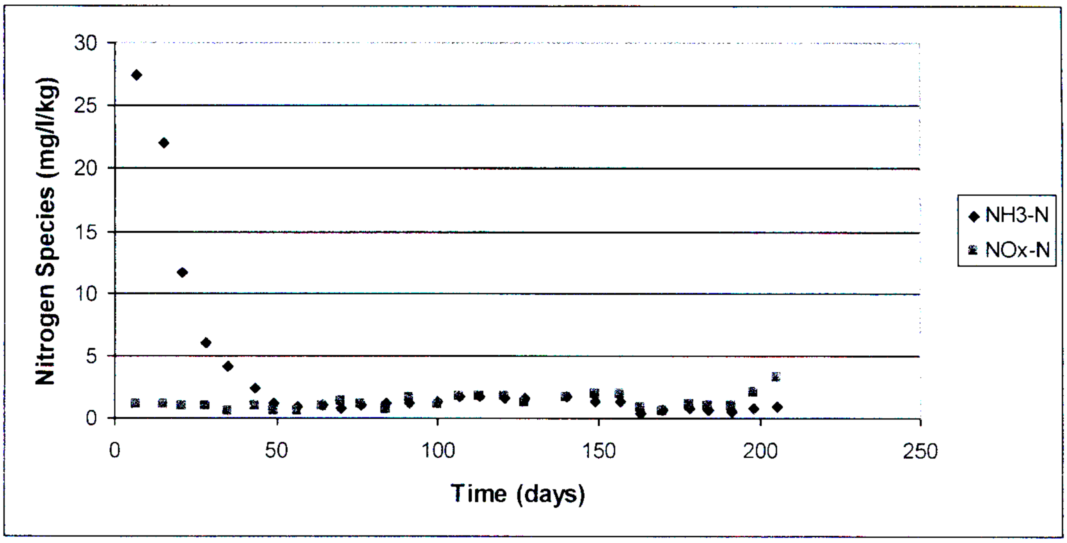

Figure 7.19 Column 2 NH_4-N and NO_x-N

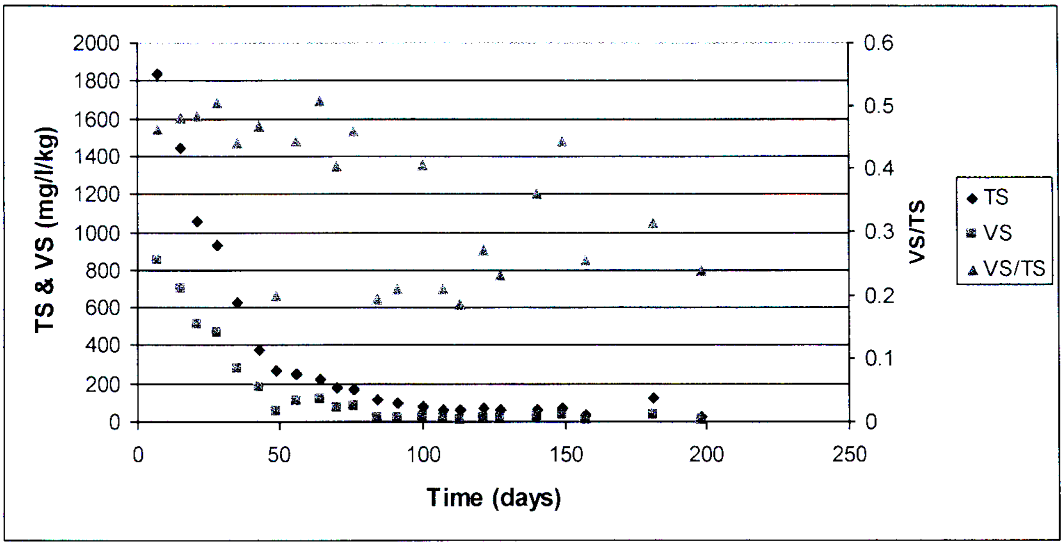

Figure 7.20 Column 2 TS and VS

The COD measured at the beginning of the test (1350mg/l/kg) drops rapidly for the first 70 days to a concentration of 50mg/l/kg (Figure 7.18). Thereafter the decrease is slower with a value of 13mg/l/kg recorded at the end of the test.

The predominant nitrogen compound in the early stage of the test is ammoniacal nitrogen with an initial value of 27mg/l/kg which drops to 1mg/l/kg over the first 50 days (Figure 7.19). Thereafter the ammoniacal nitrogen and NO_x-N concentrations are approximately equivalent with levels that do not exceed 2mg/l/kg, although a trend is noted from day 50 to 150 where the concentrations appear to rise slightly in both species. As with the COD, the majority of the ammoniacal nitrogen is removed after 50 days of leaching.

The TS and VS both show an exponential decline with 1840mg/l/kg (TS) and 840mg/l/kg (VS) at measured at the start of the test and 25mg/l/kg and 6mg/l/kg measured at the end for TS and VS respectively (Figure 7.20). Although the relation is erratic, VS/TS shows a general decrease with time as the degradable (volatile) compounds in the leachate decrease.

7.4.3 Column 3

The results of the 16 weeks treated coarse fraction tested in Column 3 are presented in Figures 7.21 to 7.24 and discussed.

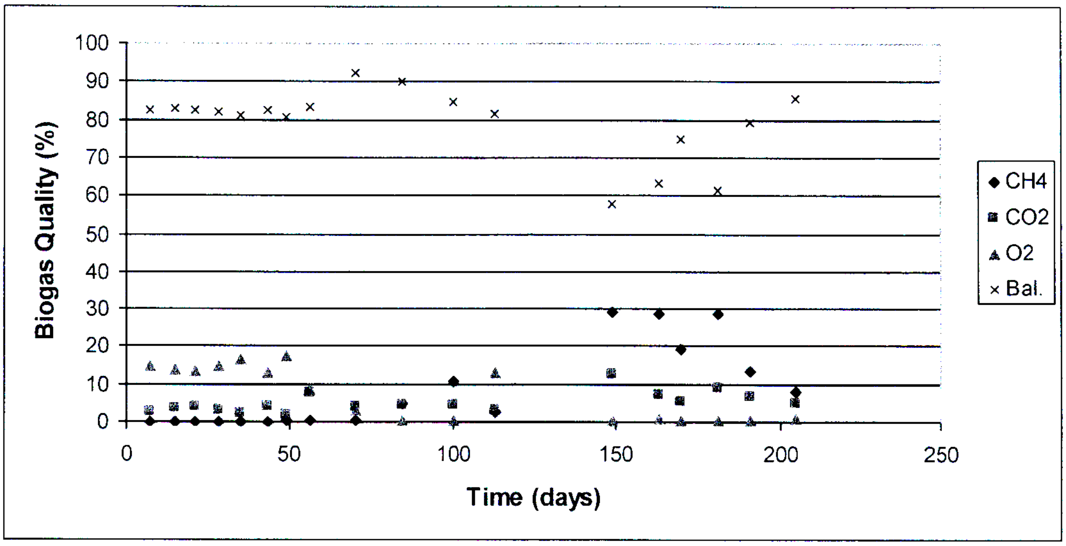

Figure 7.21 Column 3 Biogas Quality

The biogas results for column 3 shows an early increase of carbon dioxide but little methane with significant levels of oxygen (5%-15%) for the first 100 days of the test (Figure 7.21). This may be due to the column operation in which ambient air was draw into the columns during the leachate extraction. As a mitigation measure, the sampling frequency was adjusted to every two weeks, however the oxygen levels remained high and after 100 days, nitrogen was used to displace the liquid during the leachate sampling. This prevented the ingress of oxygen and ended the inhibition of the methanogenic bacteria and the methane levels rose to a maximum of 30% after 150 days and decrease thereafter, indicating a slowing down of the biological activity.

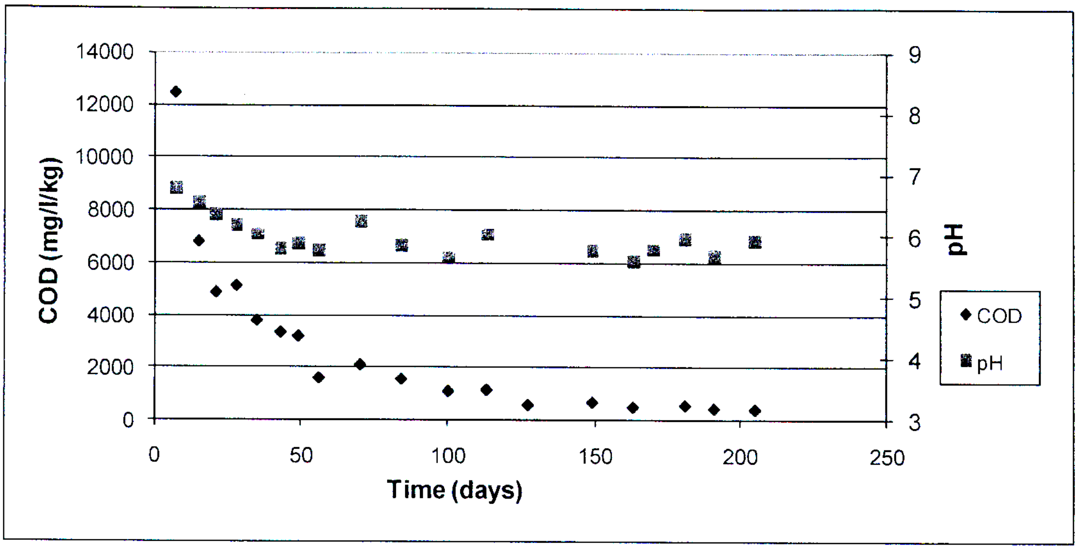

Figure 7.22 Column 3 pH and COD

The pH remains in the range of slightly above and slightly below 6 throughout the test. These pH readings are similar to column 2 which became methanogenic at an early stage, thus confirming that the methanogenic inhibition is due to the oxygen drawn into the column before day 100, and not as a result of acidic inhibition (Figure 7.22). The COD shows an exponential-like decay from an initial value of 1660mg/l/kg to a final value of 57.7 mg/kg/l (Figure 7.22).

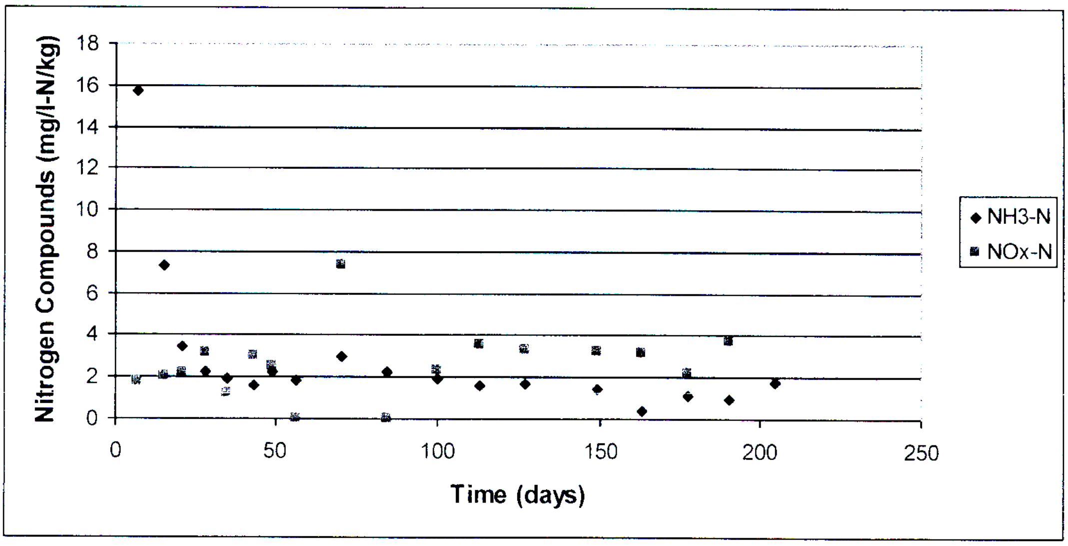

Figure 7.23 Column 3 NH_4-N and NO_x-N

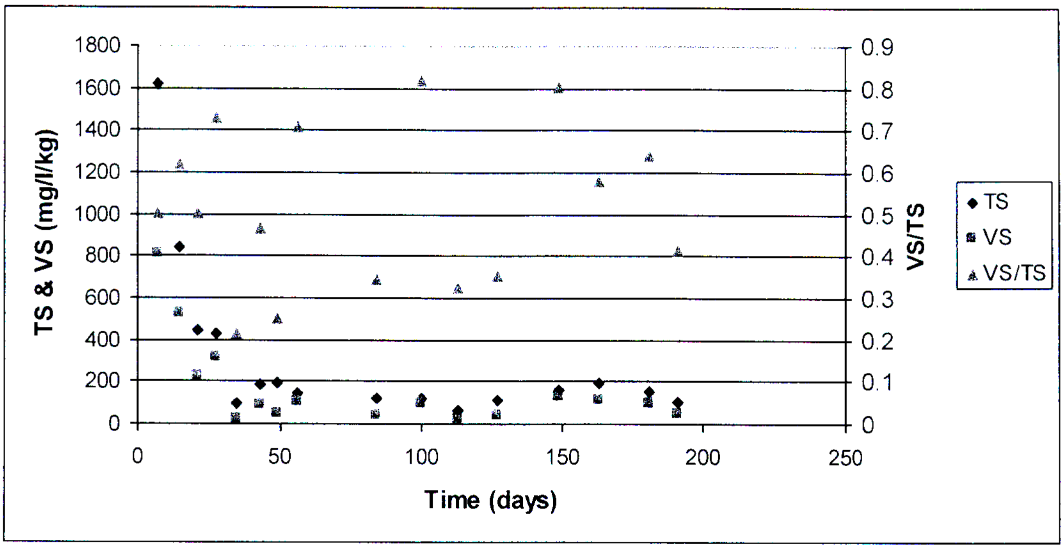

Figure 7.24 Column 3 TS and VS

The nitrogen also shows a dominance of ammoniacal nitrogen at the start of the test, but these concentrations reduce from the initial value of 16mg/l/kg to values equivalent to the NO_x-N concentrations of approximately 4mg/l/kg for the remainder of the test (Figure 7.23).

The TS and VS recordings indicate a rapid decrease from the initial respective values 1600mg/l/kg and 800mg/l/kg to a relatively stable reading of less than 200mg/l/kg and 100mg/l/kg for TS and VS respectively (Figure 7.24). The VS/TS results do not follow any clear trend over the testing period.

7.4.4 Column 4

The results of the 8 week fine material tested in Column 4 are presented in figures 7.16 to 7.20 and discussed.

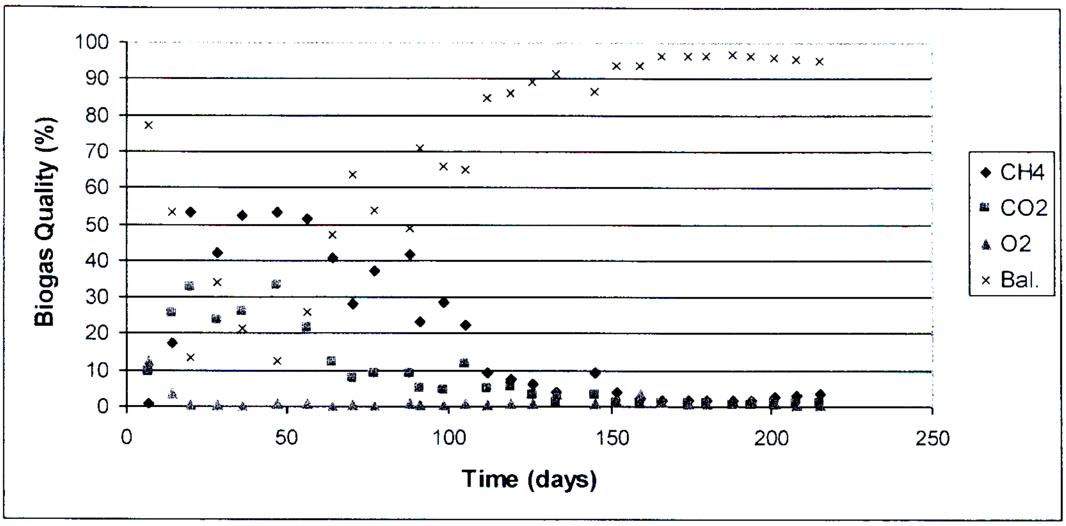

Figure 7.25 Column 4 Biogas Quality

The methane detected in column 4 rises rapidly to a maximum of 55% within 20 days from commencement, with carbon dioxide levels reaching a maximum of 35% in a similar time period (Figure 7.6). From day 50, the biogas levels drop rapidly to low levels after 150 days reflecting the decrease in biogas generation. The rapid development of methanogenic conditions, coupled with the pH which rises slowly from 6 to 7 during the test indicates that, as expected with treated waste, there was no acidic inhibition, and that the significant drop in biogas generation was not pH related (Figure 7.26). Oxygen, once consumed during the early stage of the experiment, is negligible throughout the test. The COD concentration decreased, from an initial value of 800mg/l/kg with a final measurement of 5mg/l/kg recorded on day 215.

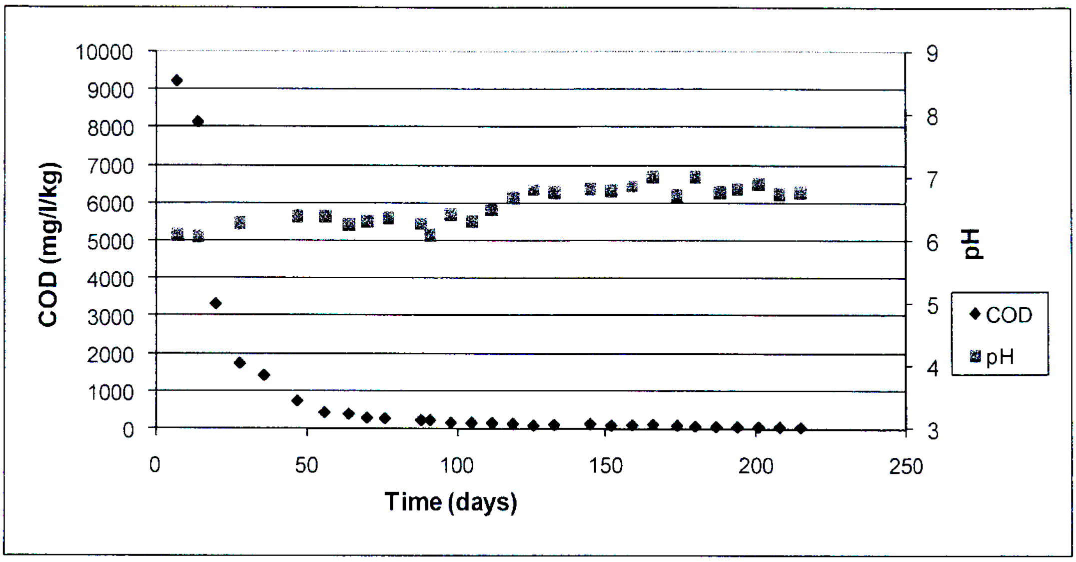

Figure 7.26 Column 4 pH and COD

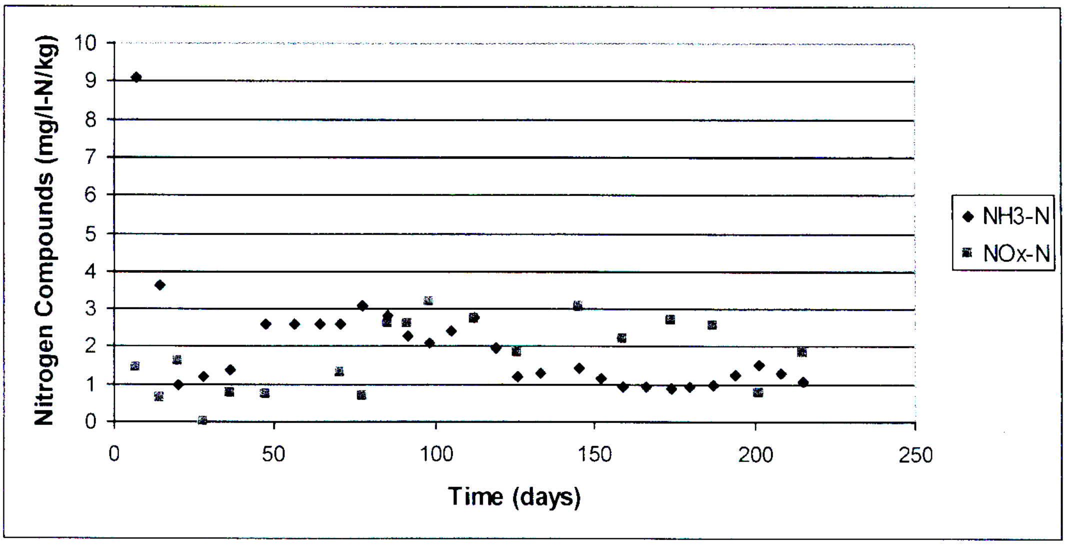

Figure 7.27 Column 4 NH_4-N and NO_x-N

The ammoniacal nitrogen was measured as 9mg/l/kg at the start of the test, but drops to 1mg/l/kg after just 20 days (Figure 7.27). Thereafter the ammoniacal nitrogen rises again to approximately 3mg/l/kg from day 50-100, before dropping to 1mg/l/kg, which is reached on day 150 and is maintained until the end of the test. The NO_x-N rises from approximately 1mg/l/kg to 3mg/l/kg (reached on day 100) and then decreases slightly over the remainder of the test, with a value of 2mg/l/kg recorded at the end.

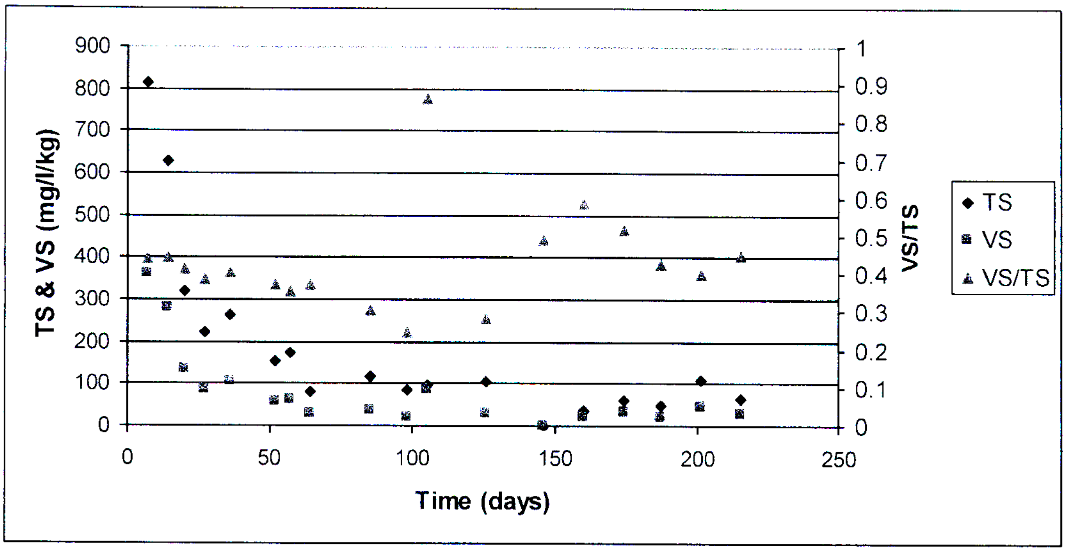

Figure 7.28 Column 4 TS and VS

The TS and VS decrease rapidly for the first 100 days from initial respective values of 820mg/l/kg and 360mg/l/kg. The readings then remain approximately 100mg/l/kg for TS and 50mg/l/kg for VS. The proportion of VS to TS appears to decrease initially from the start to day 125, but then rises again before once again decreasing (Figure 7.28).

7.4.5 Column 5

The results for the analysis on the 8 week treated coarse fraction tested in Column 5 are presented in figures 7.29 to 7.32 and discussed.

The methane levels in this column containing treated waste also rise rapidly, with methane and carbon dioxide peaking at 55 and 30% respectively after 50 days (Figure 7.29). Thereafter, the biogas concentrations drop, steadily during the course of the test, as would the biogas generation rate. The absence of an acidic stage is confirmed by the pH which remains greater than 6 for the duration of the experiment (Figure 7.30). The COD, initially 1600mg/l/kg decreases to the final COD measured as 5mg/l/kg on day 215 (Figure 7.30).

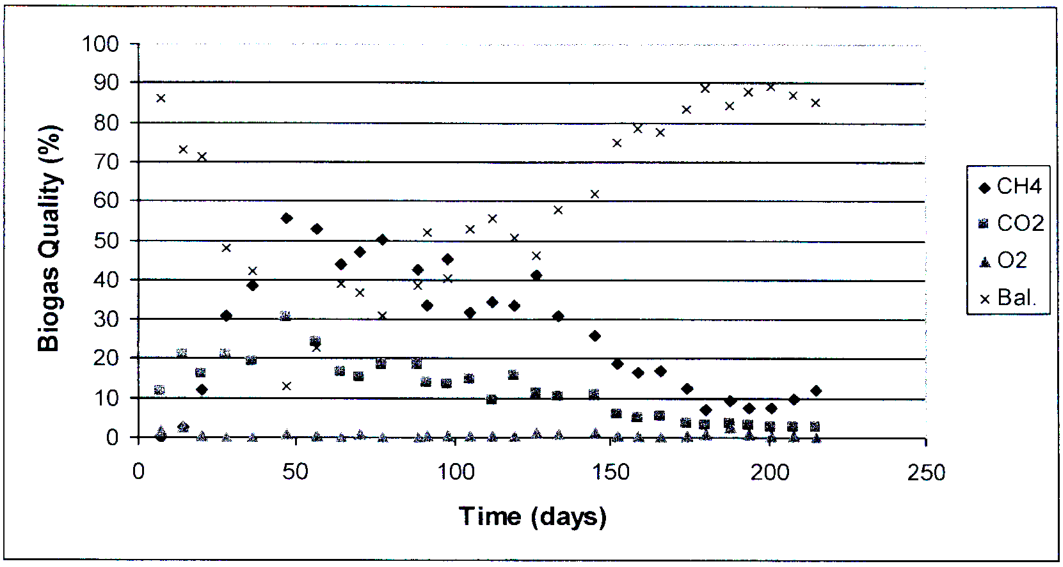

Figure 7.29 Column 5 Biogas Quality

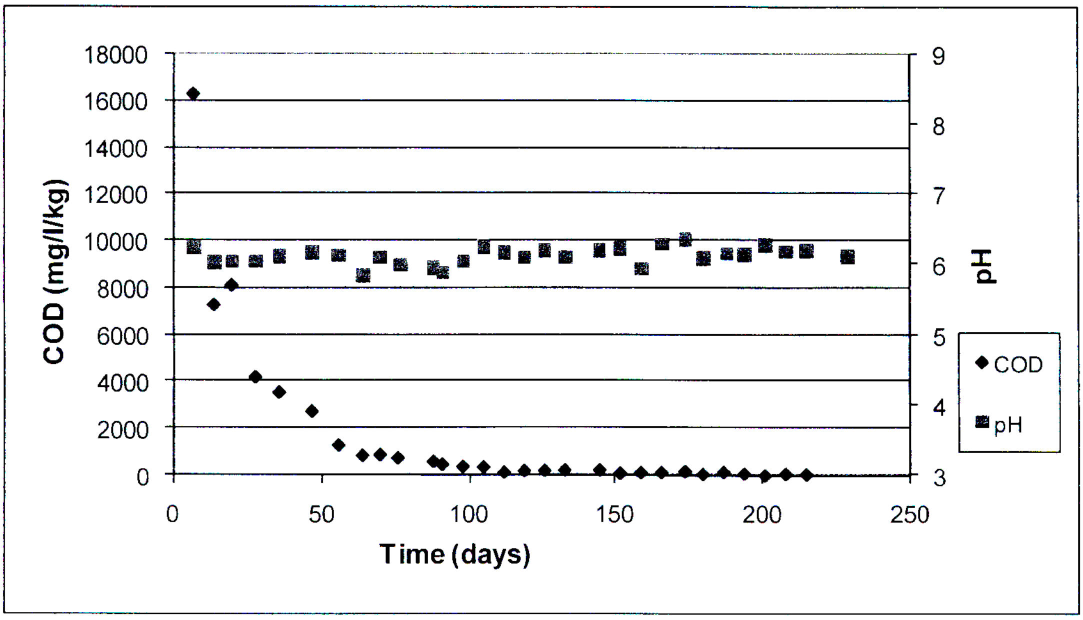

Figure 7.30 Column 5 pH and COD

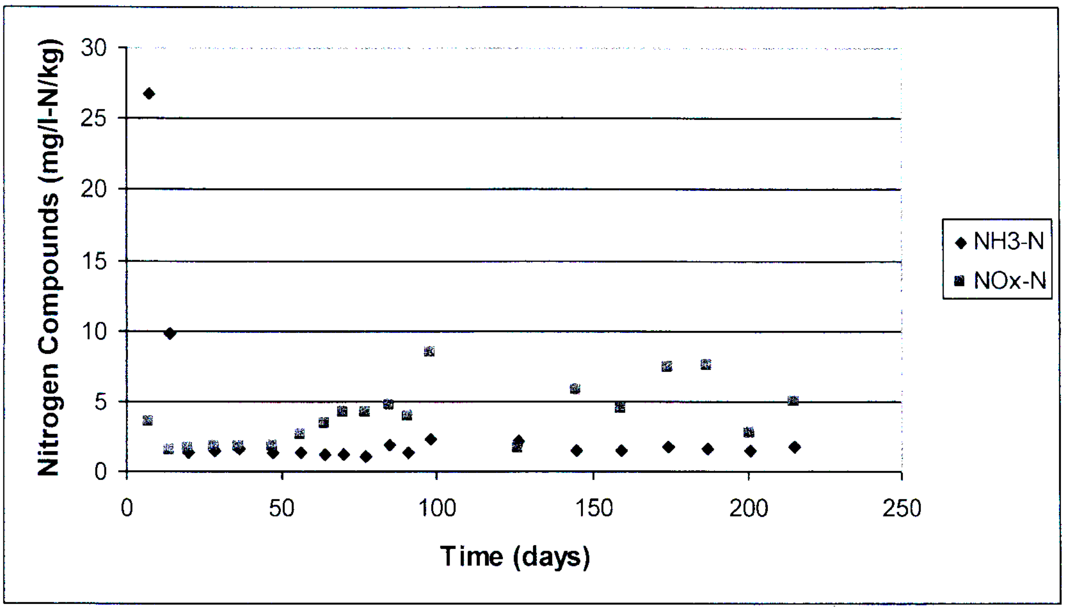

Figure 7.31 Column 5 NH_4-N and NO_x-N

Ammoniacal nitrogen was recorded in relatively high concentrations at the start of the test (27mg/l/kg) but quickly reduce to a stable range of 1-2mg/l/kg. The NO_x-N, 1mg/l/kg at first, rises to approximately 8mg/l/kg during the 215 days of testing (Figure 7.31).

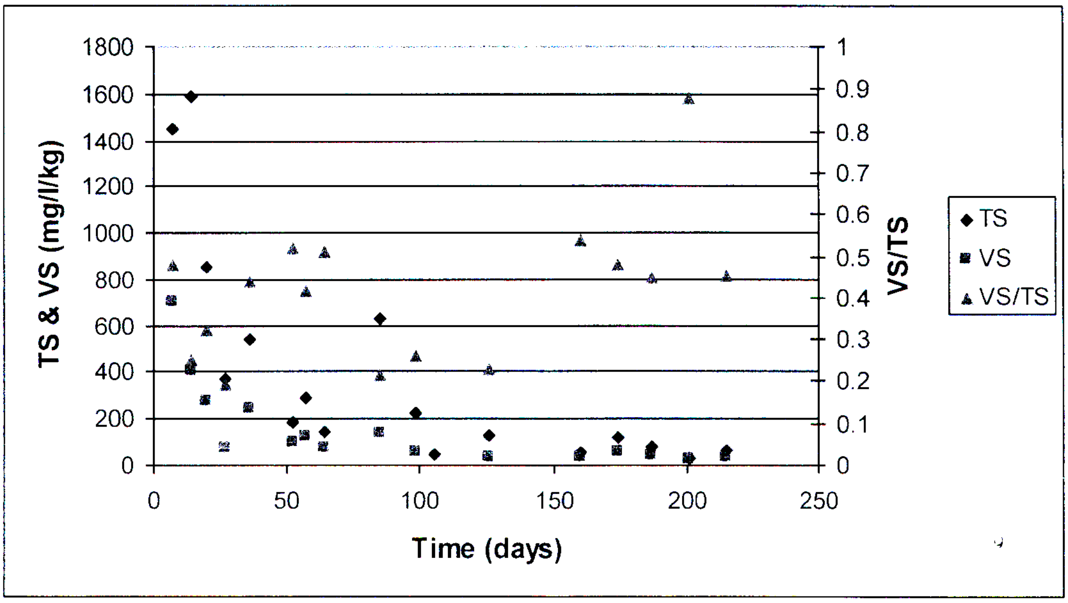

Figure 7.32 Column 5 TS and VS

The maximum TS and VS, measured at the early stages of the test, were 1600mg/l/kg and 700mg/l/kg respectively, dropping to 220mg/l/kg (TS) and 60mg/l/kg (VS) after 100 days (Figure 7.32). Thereafter, the measurements decrease more slowly to final recorded values of 63 and 28mg/l/kg for TS and VS respectively. There does not appear to be any trend in the relation between VS and TS.

7.4.6 Column Comparisons and Discussion

A major difference between the columns containing treated waste and the untreated waste column is the acidic inhibition that was only observed in the latter. This resulted in the retardation of the development of methanogenic bacteria for the first 100 days of the test, whereas the methanogenic bacteria in the other columns developed almost immediately, with the exception of Column 3, which was subject to oxygen inhibition. Furthermore, Column 1 containing the untreated waste remained biologically active for far longer than the treated waste columns suggesting a longer timeframe of release of organics.

Due to the differences in the liquid flux in the four columns containing the treated, screened waste, the comparison between the various treatment methods can only be made by assessing the leaching behavior of the waste, as presented in Chapter 8.

7.5 LYSIMETER STUDIES - PHASE 2

Three lysimeter studies were conducted on the same material tested in the leaching columns in order to assess the effect of scale and open circuit un-saturated liquid flows. The lysimeter studies were operated at a much lower relative liquid flux than the columns, resulting in a lower L/S variation and the operation of the lysimeter water balance results in a similar L/S in similar time frames between the three tests allowing for a more direct comparison of the results.

Lysimeter 1 – Untreated Waste

The results of the untreated waste lysimeter are affected by a forced raising of the pH from day 343 to day 415 by the addition of sodium hydroxide. This was done in order to accelerate the transition of the system to methanogenic conditions as the timeframe for analysis was limited and after almost one year of testing, acidic conditions were still well established. The practice of sodium hydroxide addition, also performed by Novella (1995) allowed for the assessment of the effect of methanogenesis on leachate quality. The results of the analysis on Lysimeter 1 are presented in Figures 7.33 to 7.36. In total, 480g of sodium hydroxide was added over a 72 day period.

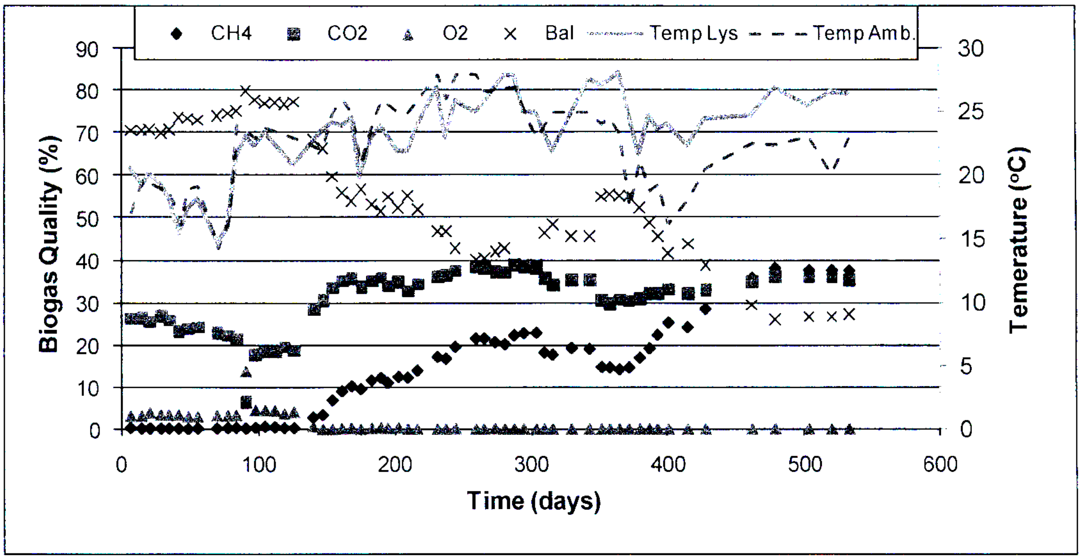

Figure 7.33 Lysimeter 1 Biogas Quality

The results from lysimeter 1 show a typical trend for untreated waste, with a protracted acidic trend. The presence of methane from day 140 onwards (Figure 7.33), despite pH levels of approximately 5.5 (Figure 7.34) shows that, as with Column 1, there must have been zones favorable to the development of the methanogens, although the system as a whole was still acidic. The addition of sodium hydroxide is clearly shown by the sharp increase in pH to neutral levels, corresponding to a further increase in methane levels. This resulted in an increase in methane levels with the decrease in nitrogen indicating an increase in biogas production. Although the dosing of sodium hydroxide was stopped on day 387, the system remains neutral as the tipping point to methanogenic conditions was reached. The temperature was affected by the ambient seasonal changes until the addition of the thermal blankets on day 343 where the lysimeter temperature remained above 20°C despite low ambient temperatures. The COD shows high sustained pollutant loading of the order of 250 mg/l/kg through the acidic phase. The COD levels drop rapidly as the pH and methane levels increase with leachate COD dropping below 50mg/l/kg at the end of the test (Figure 7.34).

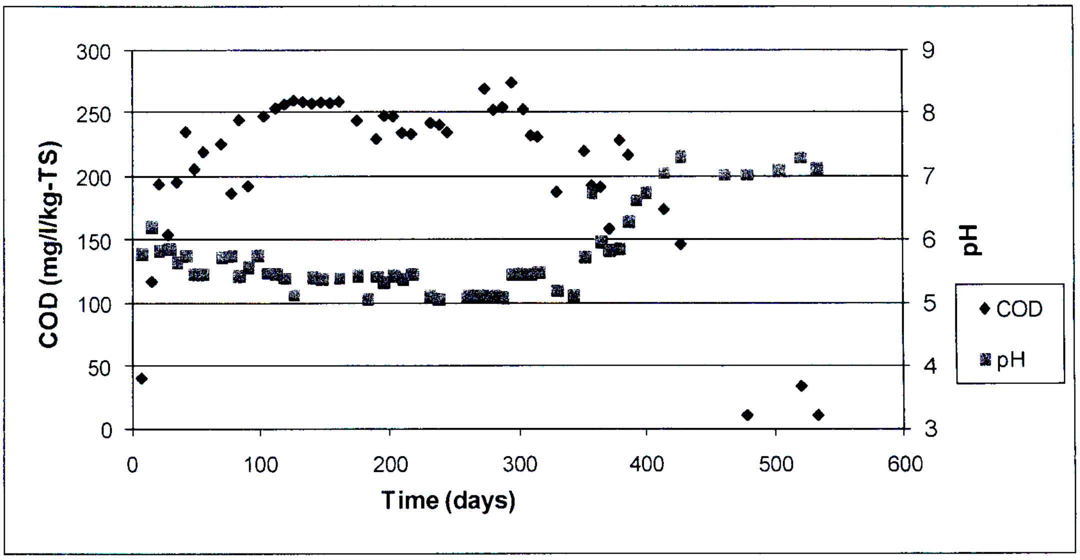

Figure 7.34 Lysimeter 1 pH and COD

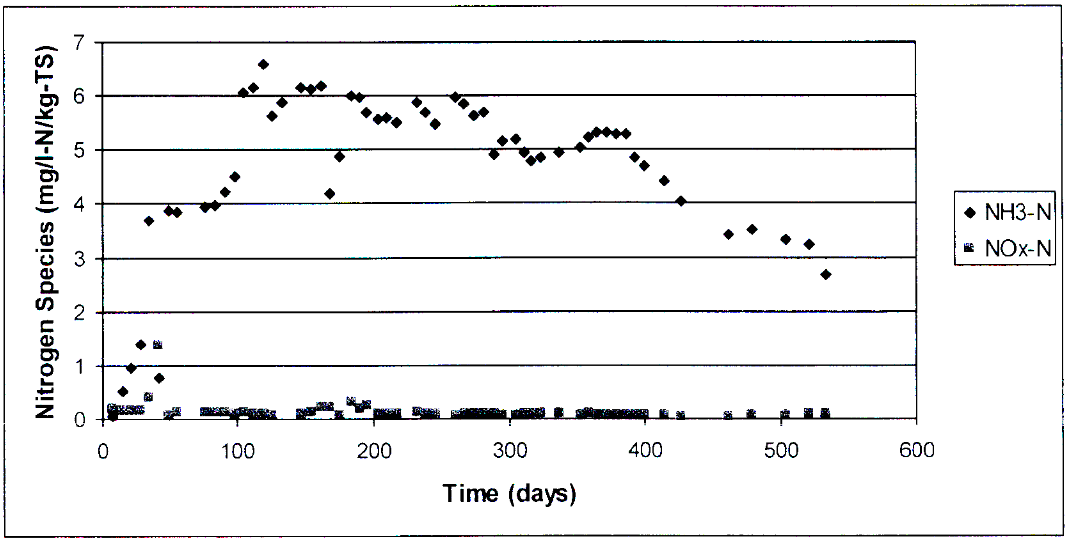

Figure 7.35 Lysimeter 1 NH_4-N and NO_x

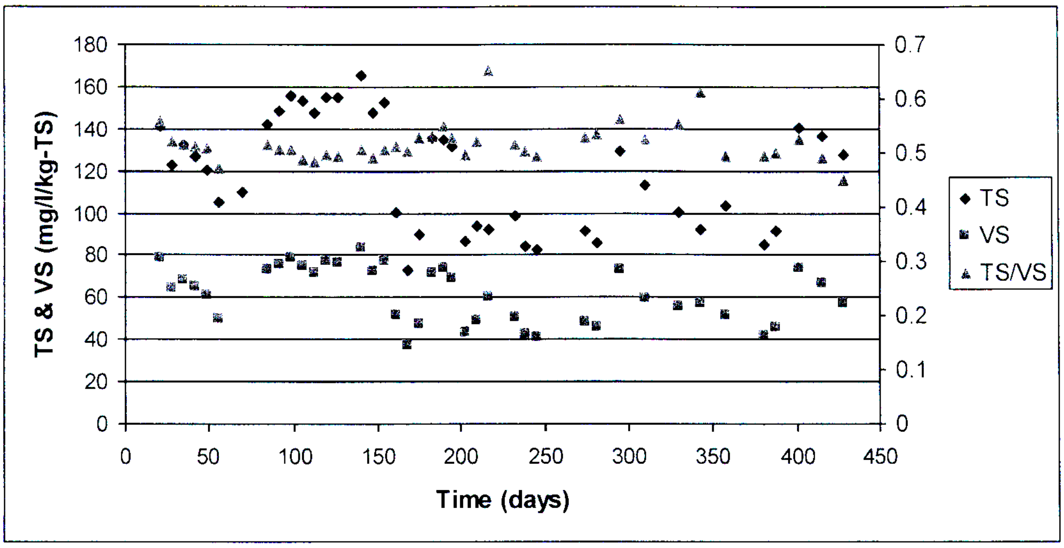

Figure 7.36 Lysimeter 1 TS and VS

Ammoniacal nitrogen levels rise for the first 180 days to a maximum value of 6.5mg/l/kg-TS, and gradually drop thereafter. A potential mechanism of removal is the flux of leachate through the lysimeter as there is no anaerobic consumption of ammoniacal nitrogen (Figure 7.35) – an assumption that is supported in the results of the leaching analysis in chapter 8. NO_x-N was detected at low levels throughout the test. The scattered TS and VS results appear to follow a relatively flat trend with values ranging from 70-160mg/l/kg-TS for the TS and 40-80mg/l/kg-TS for the VS. The TS/VS ratio of approximately 0.5 was maintained throughout the test (Figure 7.36).

Lysimeter 2

The results for Lysimeter 2 which contained the 16 weeks treated fine fraction are presented in Figures 7.37 to 7.40.

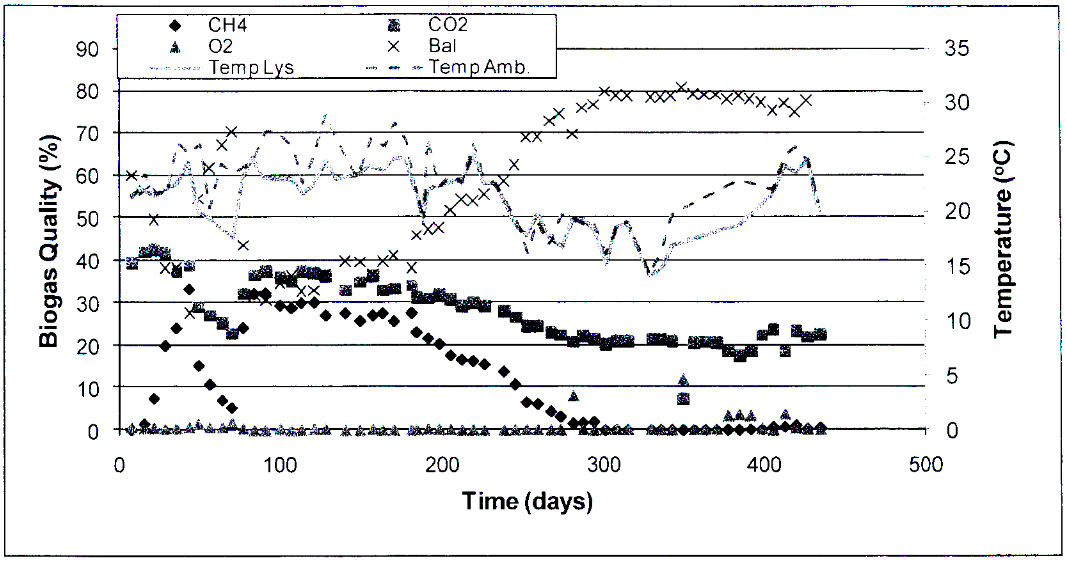

Figure 7.37 Lysimeter 2 Biogas Quality

The biogas quality shows an early development of methanogenic bacteria, with the methane concentration peaking after approximately 100 days. The methane concentration then decreases to negligible levels after 300 days (Figure 7.37). The levels of nitrogen in the lysimeter headspace increase from day 200 onwards, showing a decrease in the biogas generation rate. The temperature in the lysimeter is clearly affected by the ambient seasonal temperature changes and the decrease in methanogenic activity after approximately 250 days may be as a result of low temperatures. However, when the lysimeter temperatures increase to above 20°C the methane concentrations remain low suggesting that the drop in methane levels is not temperature related.

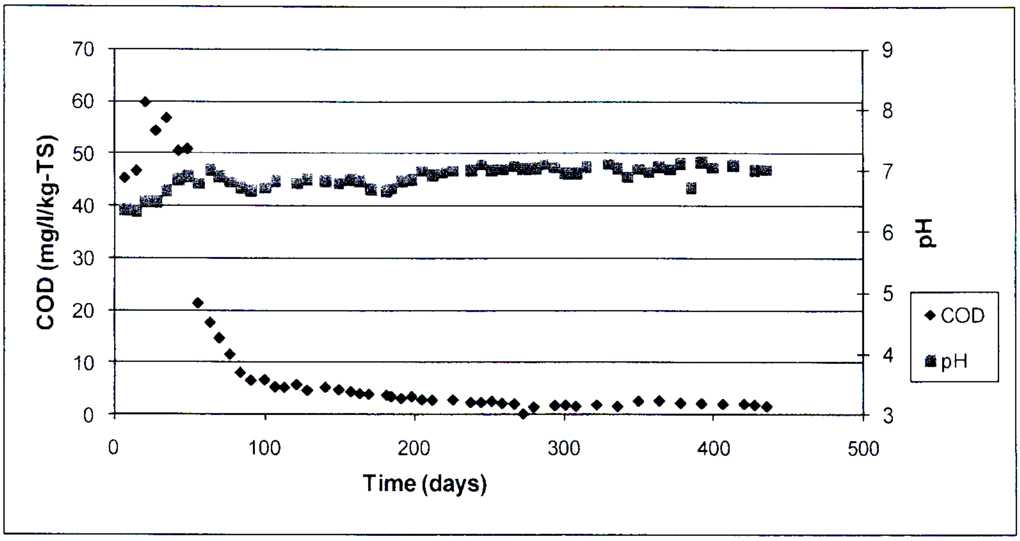

Figure 7.38 Lysimeter 2 pH and COD

The pH rises from approximately 6.5 to 7, once again showing the lack of an acidic stage for the treated waste (Figure 7.38). The COD rises rapidly from the start to a maximum value of 60mg/l/kg-TS on day 35, decreasing rapidly thereafter to approximately 6mgl/kg-TS after 100 days.

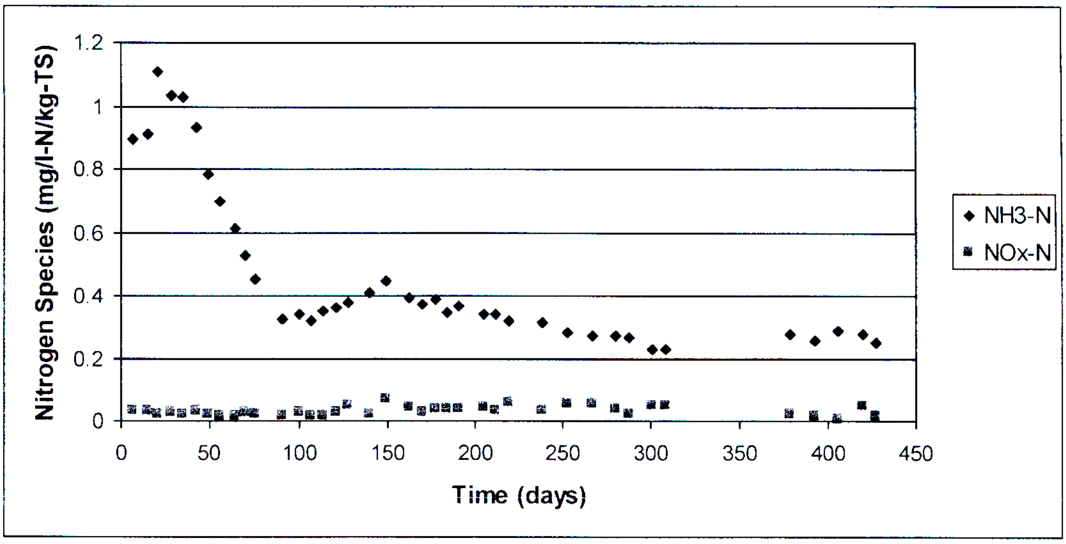

Figure 7.39 Lysimeter 2 NH_4-N and NO_x-N

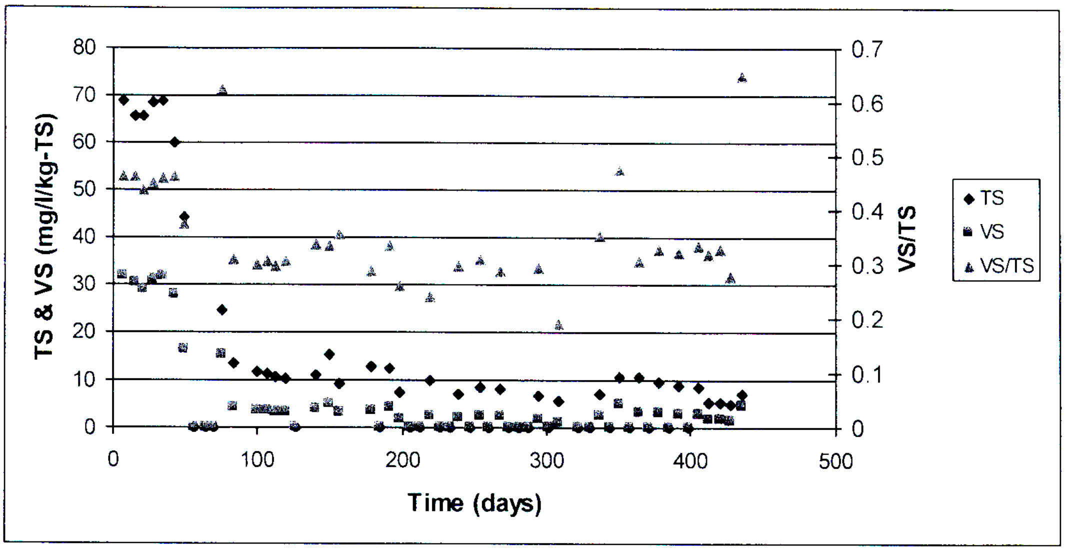

Figure 7.40 Lysimeter 2 TS and VS

The concentrations of ammoniacal nitrogen, the more prevalent species, decreases rapidly from the early peak value of 1.1mg/l/kg-TS to approximately 0.34mg/l/kg-TS on day 100. The values then show an upward trend until day 150 before once again decreasing at a slower rate (Figure 7.39). The rapid decrease of ammoniacal nitrogen measured between days 3 and 100 is probably attributed to physical removal via the leachate flux. Low concentrations of NO_x-N were detected throughout the test. The TS and VS reflect the same trend as the COD, decreasing rapidly from the start to day 100, and dropping more slowly thereafter. Maximum values measured were 69mg/l/kg and 32mg/l/kg for the leachate TS and VS respectively. The VS/TS appears to drop from the start to day 100 and then stabilizes, although the relationship is fairly erratic (Figure 7.40).

Lysimeter 3

The results for the 16 week treated coarse waste tested in Lysimeter 3 are presented in Figures 7.41 to 7.44

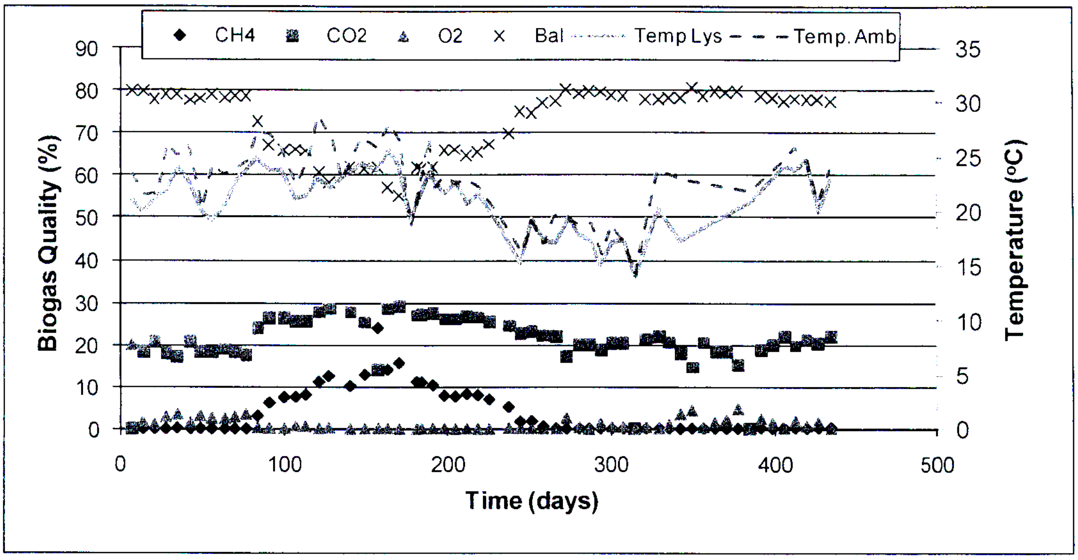

Figure 7.41 Lysimeter 3 Biogas Quality

The biogas results show a delay in the onset of methanogenic activity, which, when considering the pH, is most likely due to the presence of oxygen in the early stages of the test (Figure 7.36). The methane production begins after approximately 80 days, but does not reach very high levels, with a maximum concentration of 24% recorded on day 157. The methane production drops to negligible levels after 250 days, with the low concentration of oxygen and high carbon dioxide levels indicating that the system is still anaerobic although the pH shows that conditions were suitable to methanogenic activity throughout the testing period (Figure 7.42). The lysimeter temperatures are affected by the seasonal temperature variations and although the temperatures do drop below temperatures suitable for methanogenic conditions (20°C), the methanogenic activity does not increase with the increase in temperatures. The COD rises from 17.5mg/l/kg to a peak of 71.6mg/l/kg with 8.4mg/l/kg measured at the end of the test (Figure 7.38).

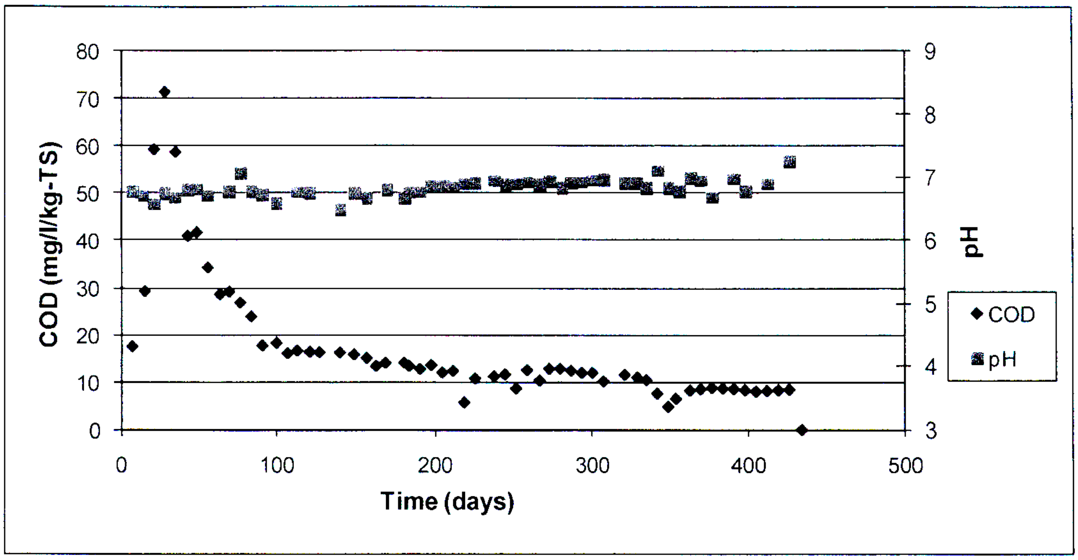

Figure 7.42 Lysimeter 3 pH and COD

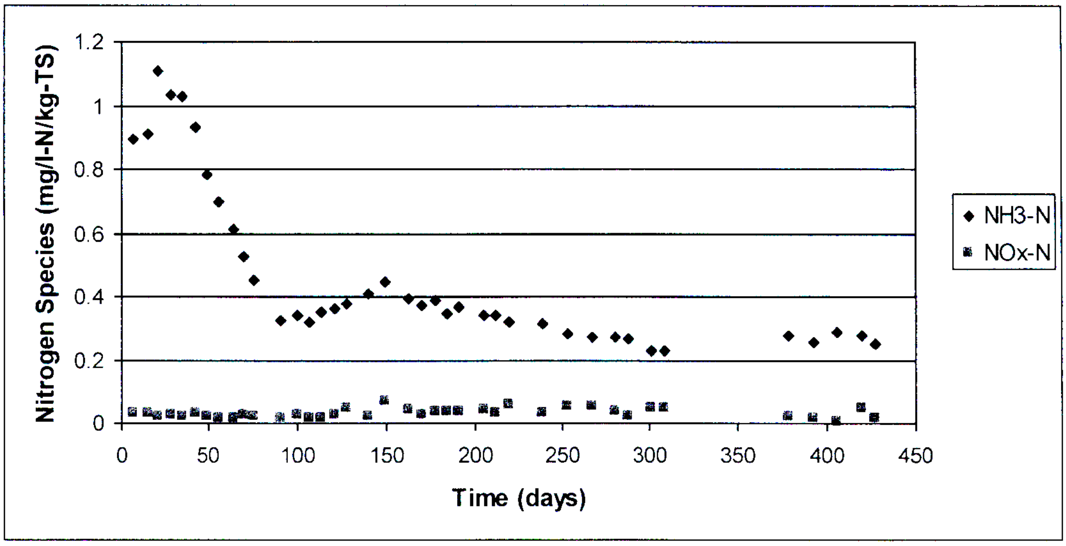

Figure 7.43 Lysimeter 3 NH_4-N and NO_x-N

The ammoniacal nitrogen decreases rapidly, rising slightly as the methane levels increase, before once again decreasing (Figure 7.43). As with Lysimeter 2, the rapid drop in ammoniacal nitrogen levels is probably due to physical removal via the liquid flux, an assumption which is supported by the analysis of the leaching discussed in chapter 8. The NO_x-N levels remain low through the test.

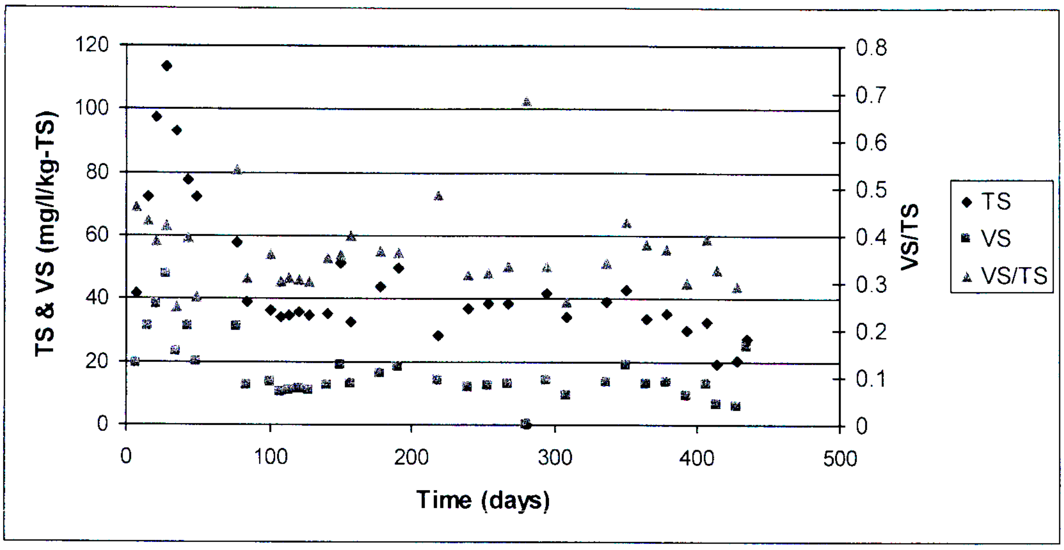

Figure 7.44 Lysimeter 3 TS and VS

The TS and VS results reflect the trends of the COD results, with the initial rapid increase followed by a rapid decrease (Figure 7.44). From day 100 the decrease is less intense. The erratic VS/TS relationship does not appear to have any distinct trend.

Comparisons

The assessment of the biogas quality show that the untreated waste, as reported in other research, was subject to an extended stage of acidic inhibition, which was absent in the lysimeters containing treated waste. The effect of treatment on leachate parameters COD and ammoniacal nitrogen is well illustrated by figures 7.45 and 7.46 where the results are plotted on the same set of axes.

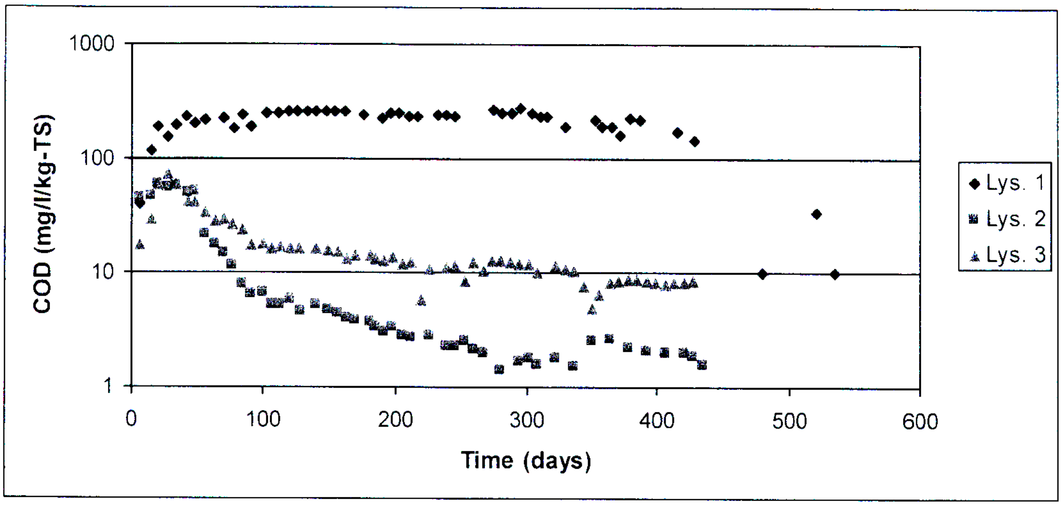

Figure 7.45 Comparison of Lysimeter COD

Note that a log scale is used due to the large variation in magnitude between the results. The COD shows clearly the benefit of treating waste, particularly in terms of the high COD load during the acidic stage, with the leachate COD concentration of the untreated waste an order of magnitude greater than that of the treated waste. However, after the establishment of methanogenic conditions, the concentration of the untreated waste COD decreases to levels approximately equal to that of the coarse treated waste fraction. It is interesting to note that the emissions from the fine waste were significantly lower than those of the coarse fractions, suggesting that the coarse fractions have a higher post landfill emission potential in the long term.

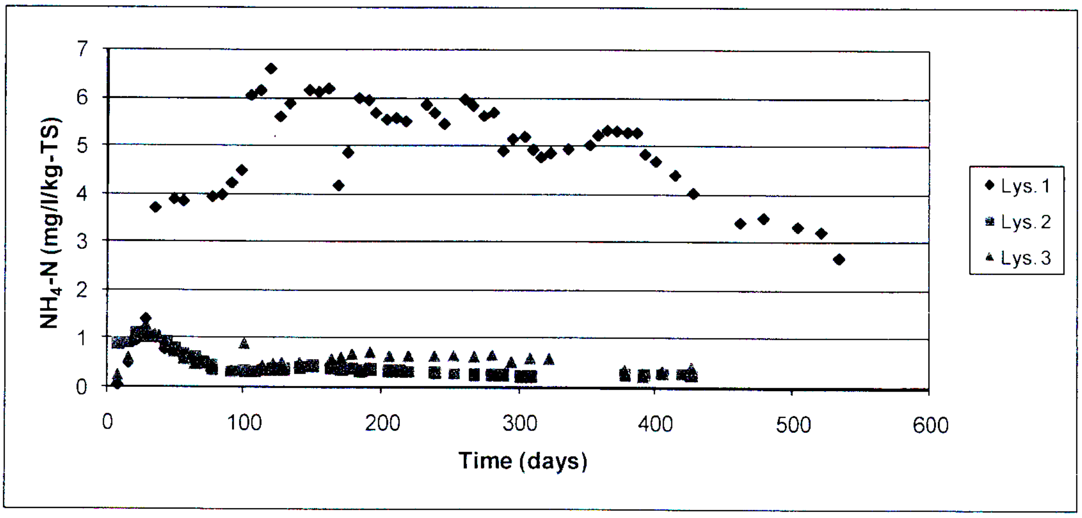

Figure 7.46 Comparison of Lysimeter NH_4-N

The untreated waste shows a much greater ammoniacal nitrogen load than the treated waste. The ammoniacal nitrogen concentration of the treated waste is quite similar, although the readings are slightly higher in lysimeter 3 which contains the coarse fractions.

Assessment of Pollutant Leaching

The results presented in this chapter provide a qualitative assessment of the emission potential. Modeling of the leaching of pollutants from the columns and lysimeters provide a further assessment of the cumulative loading and long term pollutant potential of the wastes analysed. This assessment is presented in the following chapter.

CHAPTER 8

MODEL OF THE LEACHING PROCESS AND EMISSION PRODUCTION

8.1 INTRODUCTION

As stated in Chapter 7, the direct comparison of column concentration is not beneficial due to the different degree of dilution caused by the varying porosity and differences in liquid flux. An expression is required to describe the chemical and biochemical solubilization and degradation processes that occur in conjunction with the physical removal through the liquid flux. The formulation of a mathematical model that describes the leaching process is thus needed and has numerous benefits:

- A model allows for extrapolation of the results to estimate the long term behavior for time frames that fall beyond those covered in the actual testing stage.
- Comparing the model parameters obtained for different material types provides further insight into their differences.
- Furthermore, comparing the parameters of the same material which was subjected to different scales and L/S ratios, for example, a column and lysimeter, an indication is obtained of the reliability of using these results in the assessment of a full scale landfill site.

8.2 LEACHING IN ANAEROBIC SOLID WASTE SYSTEMS

According to an extensive literature review conducted by Kouzeli-Katsiri et. al (1999), two basic approaches are used in the formation of such models. In the first approach an empirical curve is fitted to data with either time or the L/S ratio used as the independent variable. Although these models are able to describe the basic trend of a high to low concentration, they tend to be specific to the conditions of that particular test. The other approach uses a more detailed view of the system processes and attempt to qualitatively describe the biological processes that occur in the system. These processes include the solubilisation or leaching of solid material, the degradation of soluble organic material and the development of acidogenic and methanogenic bacteria. A pseudo-first order kinetic equation was used to describe the solubilisation rate while the Monod model was used to describe the degradation and growth rates. A liquid model is then used to describe the flux of fluid through the system, with a completely mixed reactor the model most commonly used (Kouzeli-Katsiri et. al, 1999). In other studies liquid models of greater complexity have been implemented such as considering the system to be a series of mixed reactors (Straub et. al, 1982a), implementation of unsaturated flow models (Straub et. al, 1982b) and an imperfectly mixed reactor with diffusive mixing (Bello-Mendoza et. al, 1998). The imperfectly mixed reactor approach considers the reactor as two split zones: the flow through zone and the retention zone. Both zones are modeled as mixed reactors, connected by diffusive mixing, with the flux of liquid through the system limited to the flow through zone. This is similar to the double porosity model proposed by Bevan et al (2003).

8.2.1 Development of a Leaching Model

A process based model was developed, based on the data from the column and lysimeter tests. The mechanisms that are active in the systems can be described by the following three processes:

- The leaching of solids into the liquid phase
- The biological conversion of materials into biogas
- The physical removal of material via the liquid flux

The leaching of material from the solid state into the liquid is either chemical (dissolution) or biochemical (hydrolysis). The mass balance in the system is governed by the following reactions:

Inputs:
Dissolution of solids into the liquid state (Ds)
Hydrolysis of solids into the liquid state (Hs)

Outputs:
Removal via water flux (Fs)
Biological conversion to biogas which is vented (Bs)

The mass balance of the system between sampling can be expressed as follows:

$$C_{i+1} \cdot V_T = C_i \cdot V_T + Ds_i + Hs_i - Fs_i - Bs_i \qquad 8.1$$

Where C_i is the sample concentration; (units of mass per volume).
C_{i+1} is the sample concentration from the following analysis;
V_T is the volume of the system; units of volume;
Ds_i, Hs_i, Fs_i, and Bs_i are the leaching mechanisms for that particular interval; (units of mass): Note that the biological degradation (Bs_i) is not applicable to ammoniacal nitrogen for which there is no biological anaerobic consumption.

The mass of the compound physically removed via sampling in the context of these tests is given by:

$$Fs_i = C_i.V_s \qquad 8.2$$

Where V_S is the volume of the leachate sample. Thus 8.1 can be rewritten as

$$C_{i+1} \cdot V_T = C_i \cdot V_T - C_i \cdot V_S + Ds_i + Hs_i - Bs_i \qquad 8.3$$

For simplicity, the parameters Ds_i, Hs_i, and Bs_i are grouped together into the single parameter, Ψ:

$$\psi = Ds_i + Hs_i - Bs_i \qquad 8.4$$

Combining 8.3 and 8.4 and rearranging the equations, Ψ (in units of mass/(mass:TS)) is given by:

$$\psi = \frac{C_{i+1} \cdot V_T}{M_{TS}} - \frac{C_i (V_T - V_S)}{M_{TS}} \qquad 8.5$$

Where M_{TS} is the mass of the dry solids. Thus, for these tests, where the liquid flux is controlled, it is possible to quantify the uncontrolled net leaching and consumption (Ψ).

The columns are assumed to be completely mixed reactors due to their small size and large liquid exchange while the lysimeters are assumed to be imperfectly mixed reactors with the size of the flow-through region estimated from data/model correlation.

8.2.2. Modeling of the Leaching Processes

By plotting the data on a logarithmic scale, it was noted that the data follow distinct linear trends. As a linear trend on a log scale is equivalent to a first order exponential function, a negative slope on the logarithmic scale indicating an exponential decay while a positive slope indicates an exponential growth. Thus sequential first order exponential formulas, as used by Kouzeli-Katsiri et. al (1999), are appropriate:

$$\psi = a_j \exp\left(b_j \cdot (t - l_j)\right) \qquad 8.6$$

Where

a_j and b_j are the parameters for the first order exponential expression

l_i is the lag time for the commencement of that particular stage

t is time

Note that the sign of the b_j parameter indicates whether or not the function is a growth (positive) or decay (negative).

By combining the leaching function (Ψ) with the liquid flux, the emission model is formed. The concentration of parameter C per unit mass of vessel material (mg/l/kg) as follows:

$$\frac{C_{i+1}}{M_{TS}} = \frac{C_i(V_T - V_S)}{M_{TS}.V_T} + \frac{a_j \exp\left(b_j \cdot (t - l_j)\right)}{V_T} \qquad 8.7$$

for stage j (for example).

8.2.3 Calibration of the Model Parameters

From 8.5, the leaching function (Ψ) is calculated for each sample interval as a function of time. The function is plotted on a log scale and the different stages are identified. These timeframes are then used to determine the a_j and b_j parameters of the exponential functions through least squares regression. This presents a limitation in the model application as $\Psi<0$, indicative of a higher net consumption of organics cannot be considered on the log scale for which Ψ cannot be zero or less. The advantage of this is simplicity in the application of the emission model, however, the model may overestimate the results.

Continuity of the Model

The independent application of least squares regression to determine the leaching function (Ψ) for different stages leads to discontinuity of the model. This is contrary to what would be expected in the real case, where a continuous transition from the one stage to the next is more realistic. In order to counter this, it was necessary to link the two stages at one point. To achieve this without constraining the leaching functions to each other, it was decided to develop the functions independently and then initiate the subsequent stage at the intersection of the two functions. This allows for the two stages to form a continuous progression but still allows for independent parameter calibration.

Application of the Emission Model

The model will assess both COD and ammoniacal nitrogen leached from the system due to the availability of data and their significance in terms of environmental impact and discharge limits. The results of the determination of the model calibration are presented in the following section.

Evaluation of the Leaching functions and Emission model

The exponential functions fitted to the data will be evaluated by means of the correlation coefficient (r) - defined as a measure of the strength of a linear relationship between two variables (x and y) and is calculated using the Microsoft Excel function as follows:

$$Correl(X,Y) = \frac{\sum (x-\bar{x})(y-\bar{y})}{\sqrt{\sum (x-\bar{x})^2 \sum (y-\bar{y})^2}} \quad 8.9$$

Where X and Y are the two sets of data analysed.

A correlation coefficient close to 1 (>0.8) is considered to be a strong correlation while a correlation coefficient less than 0.5 is considered to be a weak correlation. If the correlation coefficient is close to zero, it is assumed that there is no linear correlation. This means of evaluation will be used in the assessment of the final model.

8.3 DETERMINING COLUMN EMISSION MODEL PARAMETERS

8.3.1 Column COD

Column 1 – untreated waste

The leaching of COD in the untreated waste in column 1 (equation 8.5) is plotted on a logarithmic scale in order to identify the different leaching stages (Figure 8.1).

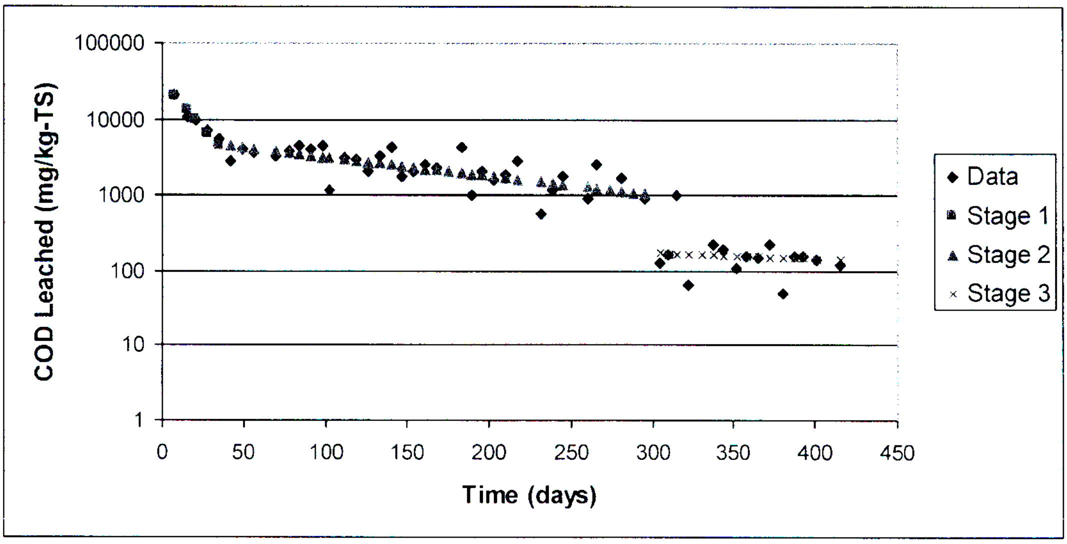

Figure 8.1 Column 1 COD Leaching Model

Table 8.1 Column 1 COD Leaching Function parameters and Emission Model Parameters

Stage	Lag (days)	a_j	b_j	Correlation
Ψ_{COD} Stage 1	0	30395	-0.055	0.95
Ψ_{COD} Stage 2	40	4654	-0.0057	0.67
Ψ_{COD} Stage 3	304	171	-0.0017	0.25
Emission Model				1.0

Three stages were identified for the column 1 COD leaching process (Figure 8.1). The first and second stage shows a strong and good correlation respectively, with the third stage discontinuous and with a poor correlation (Table 8.1). This could be due to the limited accuracy in the testing at the low end of the COD range. Furthermore, the intercept of the second and third stages would lie outside the actual range of the third stage, effectively discounting it. The transition between stage 1 and 2 and stage 2 and 3 corresponds to the onset and the peak of methane production respectively (Figure 7.10). The leaching function is combined with the liquid flux to predict the column COD concentration through the emission model (equation 8.7). This prediction is presented along with the actual COD data in Figure 8.2 and it is clear that despite the limitations of the third stage, a very good result is obtained in the prediction of the column COD with a correlation of 1.0.

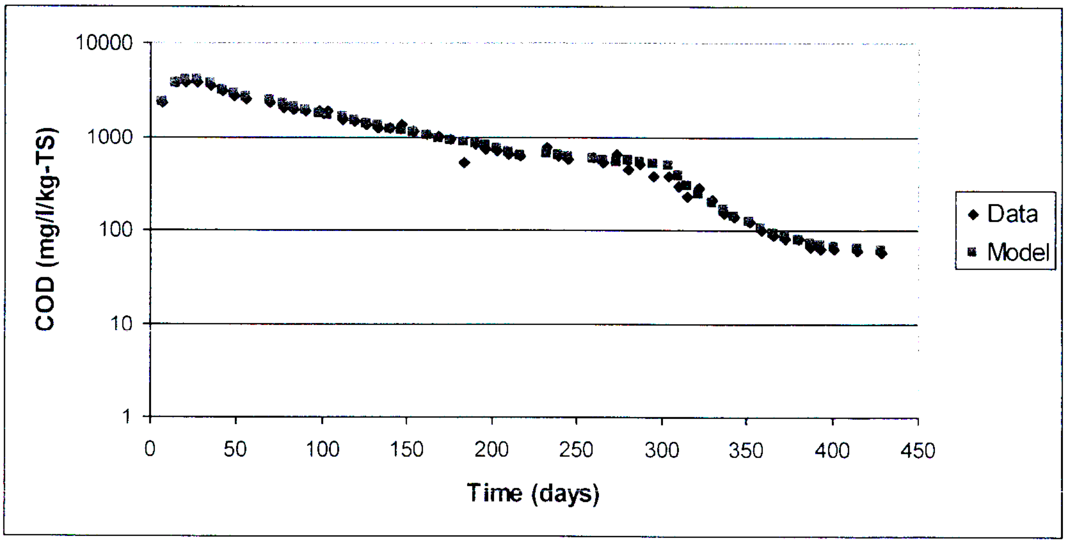

Figure 8.2 Column 1 COD Emission Model and Data Comparison

Column 2 – 16 weeks treated fine waste

The COD leached from the 16 week treated fine waste is presented on a logarithmic scale in Figure 8.3.

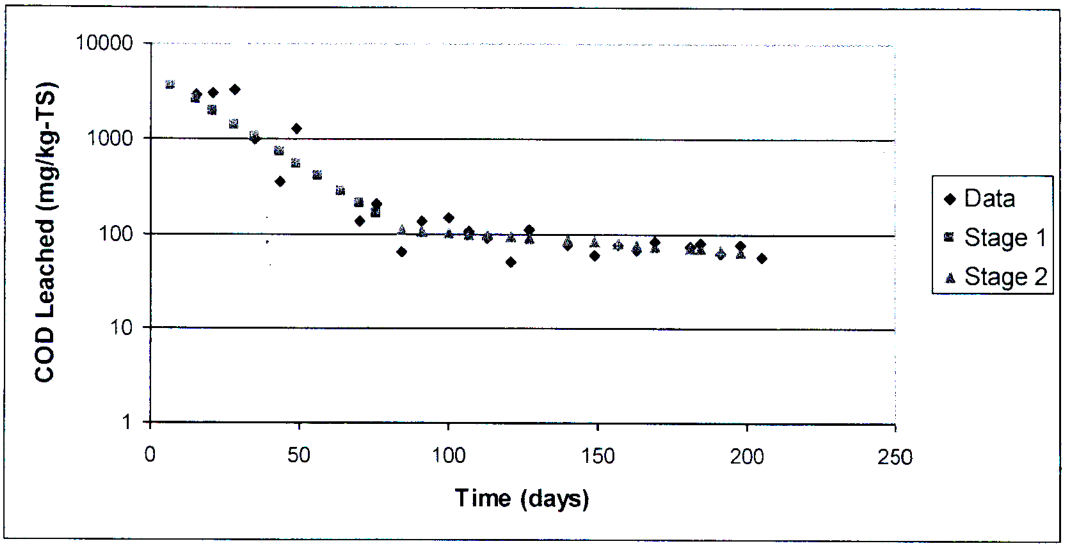

Figure 8.3 Column 2 COD Leaching Function

Two stages were identified in the solubilization processes and the model gives a very good correlation with good continuity (Table 8.2). This data is used to formulate the Emission model and despite the good correlation, the prediction of column COD is too high from days 50-100 (Figure 8.4). This is attributed to the increase in the biological removal of COD during this time frame which is not accounted for by the model. In terms of biogas activity, the transition from stage 1 to 2 corresponds to the decrease in methane generation (Figure 7.7). However, the model fits the data well from day 100 onwards and the overall correlation coefficient for the Emission model is 0.99.

Table 8.2 Column 2 COD Leaching Function parameters and Emission Model Parameters

Stage	**Lag (days)**	a_j	b_j	**Correlation**
Ψ_{COD} Stage 1	0	4965	-0.045	0.88
Ψ_{COD} Stage 2	85	109.3	-0.0046	0.71
Emission Model				0.99

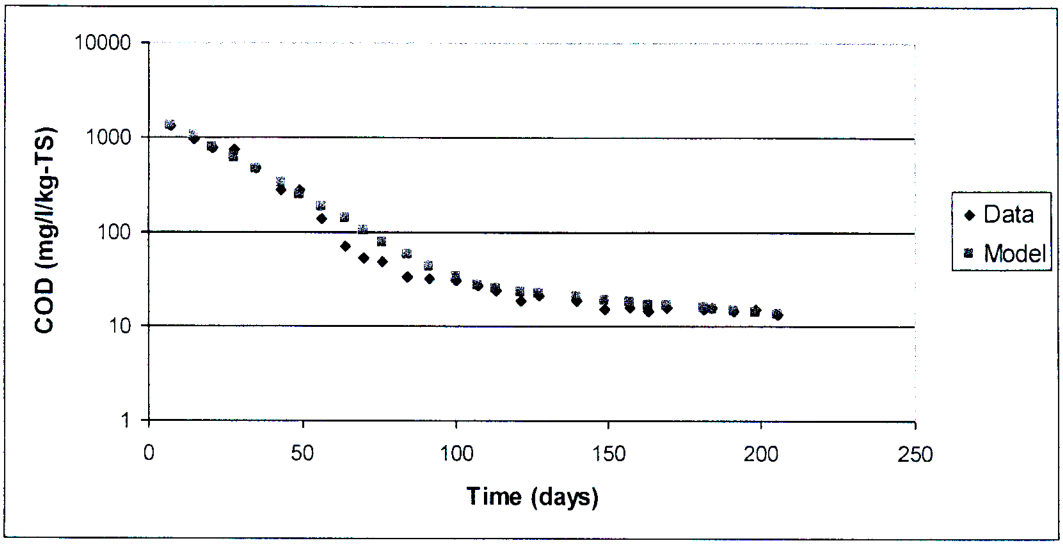

Figure 8.4 Column 2 COD Emission Model and Data comparison

Column 3 – 16 weeks treated coarse waste

The COD leached from the 16 week treated coarse waste was plotted on a logarithmic scale and two stages were identified (Figure 8.5).

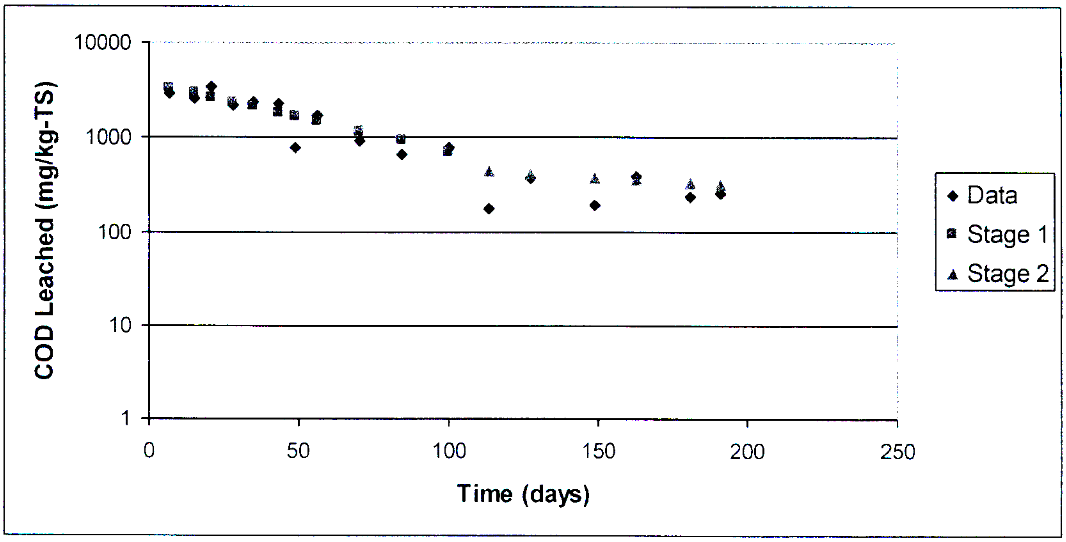

Figure 8.5 Column 3 COD Leaching Function.

A very good correlation is obtained for stage 1 for stage 2 (Table 8.3). The transition between the two stages occurs at the time when methane generation becomes more significant (Figure 7.12). With the good correlation for the leaching functions, the emission model predicts the data well (Figure 8.6) with a very strong correlation coefficient of 0.99.

Table 8.3 Column 3 COD Leaching Function parameters and Emission Model Parameters.

Stage	Lag (days)	a_j	b_j	Correlation
Ψ_{COD} Stage 1	0	3691	-0.017	0.87
Ψ_{COD} Stage 2	109	437.633	-0.0044	0.85
Emission Model				0.99

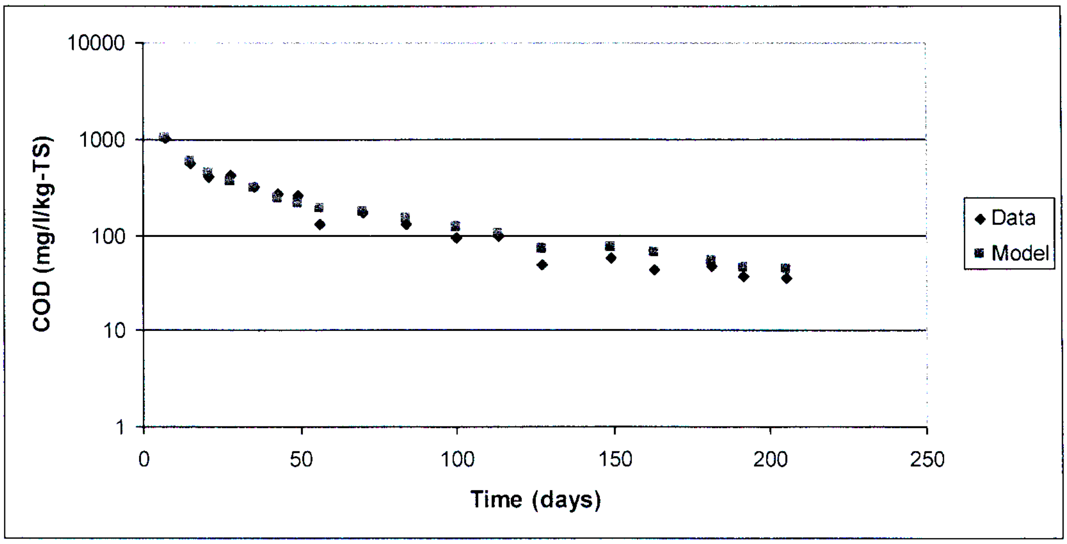

Figure 8.6 Column 3 COD Emission Model and Data Comparison

Column 4 – 8 weeks treated fine waste

The leaching of the 8 weeks fine material COD is plotted on a logarithmic scale (Figure 8.7) where two distinct stages are identified.

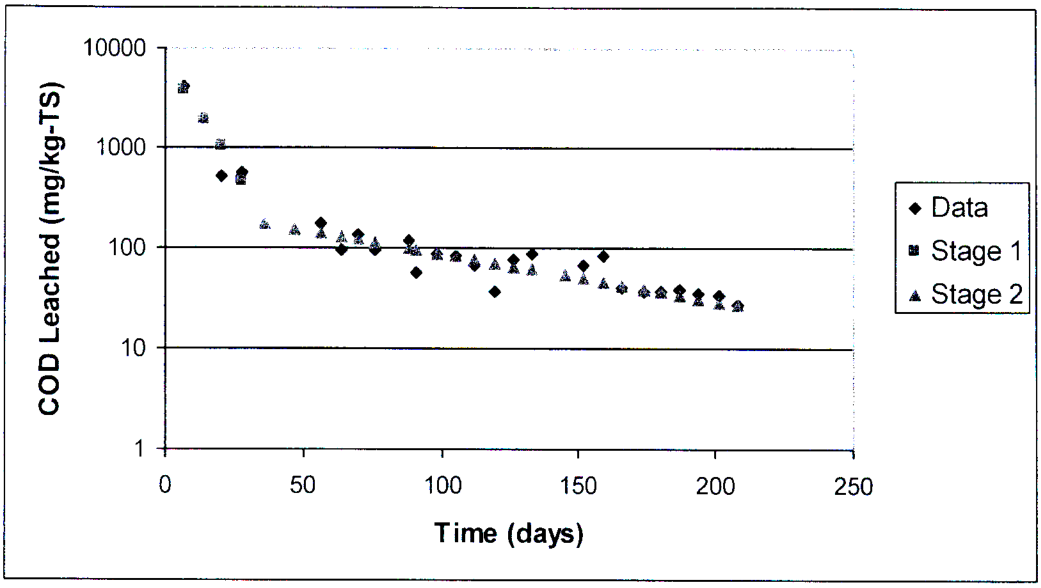

Figure 8.7 Column 4 COD Leaching Function

Table 8.4 Column 4 COD Leaching Function parameters and Emission Model Parameters

Stage	Lag (days)	a_j	b_j	Correlation
Ψ_{COD} Stage 1	0	7958	-0.1023	0.99
Ψ_{COD} Stage 2	42	165	-0.011	0.85
Emission Model				0.95

The leaching functions both show a good correlation of 0.99 and 0.85 respectively. The model parameters are presented in Table 8.4. However, the model prediction is too high in the early stages (Figure 8.8) and this may be attributed to the biological activity at the start of the test, which is not accounted for by the model. However, the tail concentrations of the model fit the test data very well (Figure 8.8) and a very strong correlation coefficient of 0.95 is calculated for the Emission model. The transition between the stages is at the time of peak methane generation as noted in Figure 7.25.

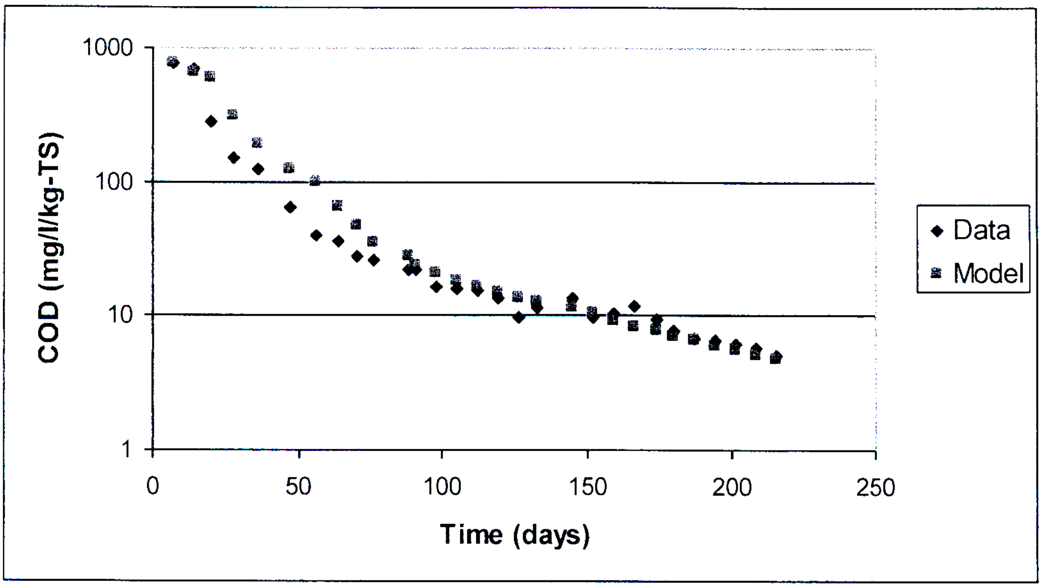

Figure 8.8 Column 4 COD model and Data comparison

Column 5 – 8 weeks treated coarse waste

The COD leached from the 8 weeks treated coarse fractions is plotted on a logarithmic scale, and although the results show a large scatter, two first order linear functions are fitted to the data (Figure 8.9).

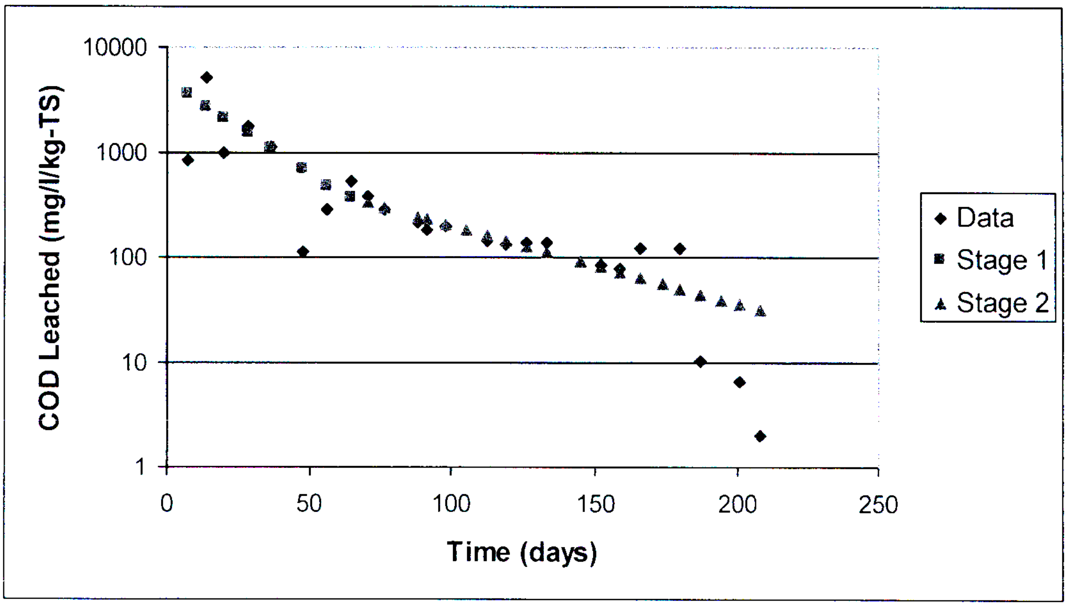

Figure 8.9 Column 5 COD Leaching Function

Table 8.5 Column 5 COD Leaching Function Parameters and Emission Model Correlation

Stage	Lag (days)	a_j	b_j	Correlation
Ψ_{COD} Stage 1	0	4861	-0.042	0.45
Ψ_{COD} Stage 2	60	400.7	-0.017	0.93
Emission Model				0.99

Stage 1 shows a weak correlation due to large variations in the leaching during this stage (Figure 8.9). Stage 2 shows a much better correlation and ultimately the Emission model does well in predicting the actual column COD (Figure 8.10) with a very strong correlation coefficient of 0.99. Peak methane levels were recorded on day 50, just before the transition from stage 1 to stage 2 (Figure 7.29).

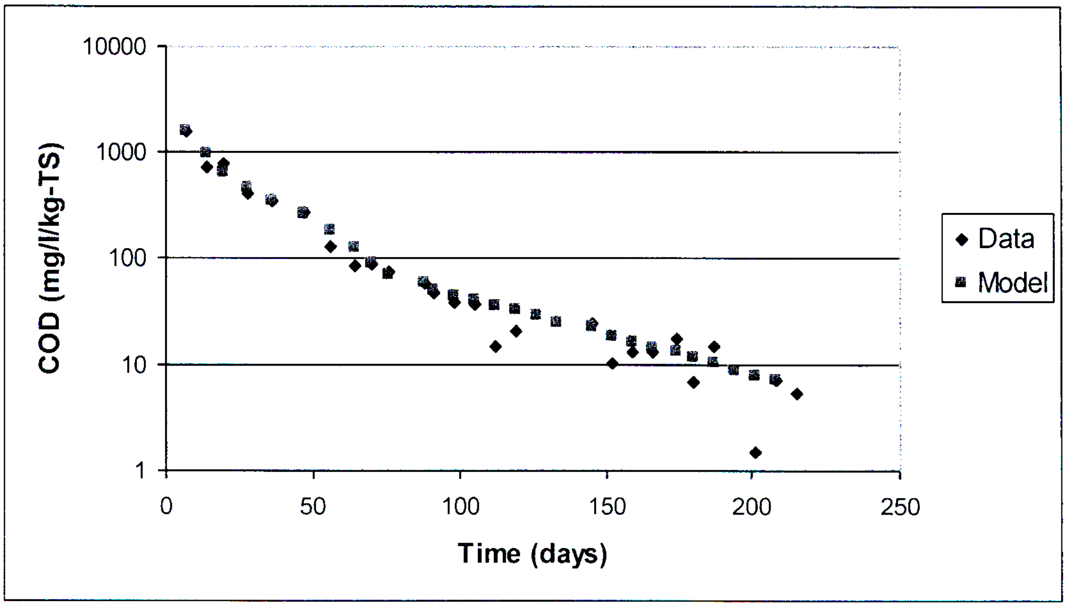

Figure 8.10 Column 5 COD Model and Data comparisons

Comments on the Emission Model for COD release

The model provides a very good correlation with the model from all the columns exhibiting a correlation coefficient of greater than 0.95. In most cases two stages were used, except for Column 1, where three stages are used. Using the model to predict the column COD works very well for the second stage where the removal processes are leachate-flux dominated and biological consumption is low. For the first stage, where the biological consumption is high, the model tends to overestimate the results as the removal of contaminants biologically ($-\Psi$) is not taken into account. However, the nature of the first-order kinetics (where higher values result in a higher differential) in conjunction with the low biological activity in the second stage appears to correct the model results. An interesting observation is the similarity in the b_1 and b_2 parameters between columns 2 and 3 and between columns 4 and 5 – columns with material sourced from the same windrow – this suggests that the leaching function may be general to waste of similar treatment timeframes.

The model provides an indication of the rate of pollutant release between the different columns. Another observation worth noting is the more rapid reduction in the rate of leaching in the fine waste columns (2 & 4) when compared to the coarse waste columns (3 & 5). The rate of decay (b parameter) of the leaching from the fresh waste column is similar to that of the treated waste columns, with the great difference being the higher initial release rate (a parameter). The transition time from the first to second stage is in all cases linked to some change in the methanogenic activity within the columns. Furthermore, considering the large differences in L/S ratios in the 5 columns, the transition does not appear to have any correlation with the L/S ratio and suggests that the change is independent of COD concentration but is rather a biologically driven process.

8.3.2 Column Ammoniacal Nitrogen

The second parameter, ammoniacal nitrogen, is modeled in the same way as COD, with the formation of the leaching function using least squares regression and evaluation of the emission model using correlation coefficients. The results for the modeling of ammoniacal nitrogen leaching in the columns are presented below.

Column 1 – untreated waste

Plotting the leaching of ammoniacal nitrogen from the untreated waste on a logarithmic scale shows three different stages (Figure 8.11).

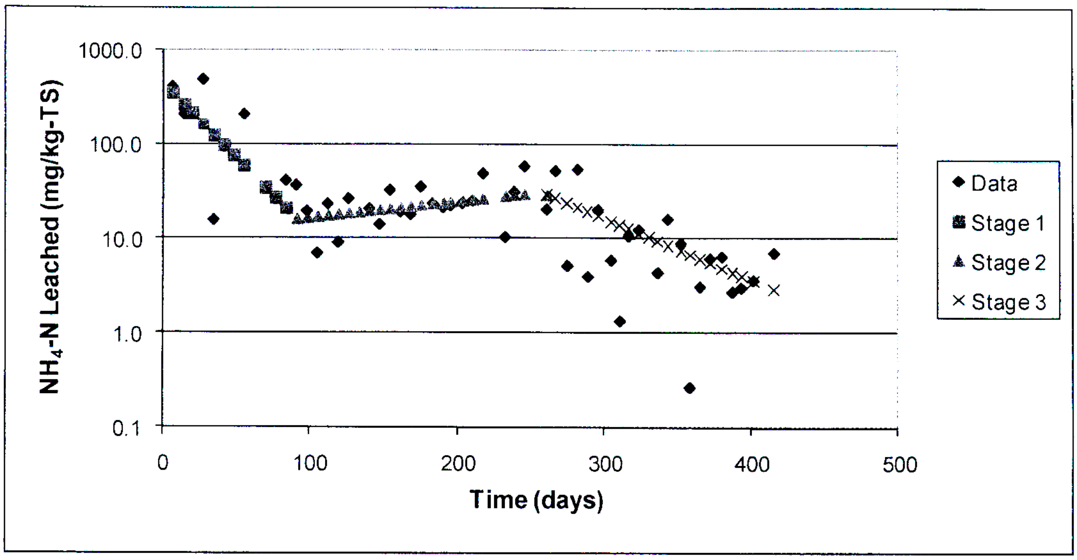

Figure 8.11 Column 1 NH_4-N Leaching Function

Table 8.6 Column 1 NH_4-N Leaching Function parameters and Emission Model Parameters

Stage	Lag (days)	a_j	b_j	Correlation
$\Psi_{NH4\text{-}N}$ Stage 1	0	456	-0.036	0.66
$\Psi_{NH4\text{-}N}$ Stage 2	90	16.28	0.004	0.68
$\Psi_{NH4\text{-}N}$ Stage 3	255	31.9	-0.0015	0.79
Emission Model				0.90

The first stage, an exponential decay, is likely to be due to the rapid leaching of readily available ammoniacal nitrogen, while the second shows an exponential growth in the ammoniacal nitrogen leached due to increased biological activity. The third stage shows a decrease in the leaching as available ammoniacal nitrogen decreases due to the biological consumption of the column substrate.

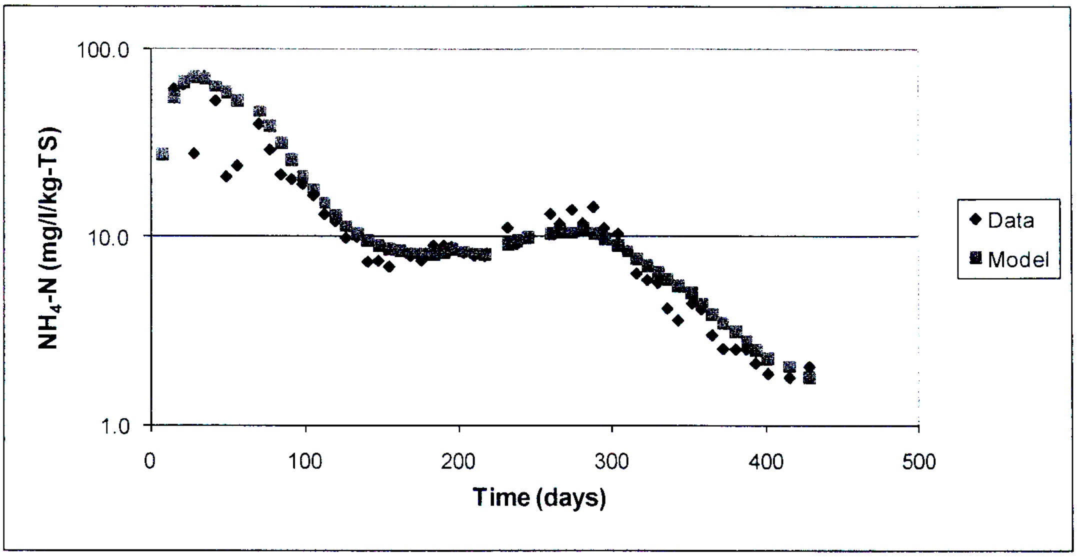

Figure 8.12 Column 1 NH_4-N Model and Data Comparison

While the correlation coefficients of the functions for the three stages are fair (Table 8.6), the emission model provides a good prediction of column ammoniacal nitrogen concentrations with a correlation coefficient of 0.90 (Figure 8.12).

Column 2 – 16 weeks treated fine waste

In the assessment of the 16 week fine material, three stages are again identified from the logarithmic plot (Figure 8.13). As for column 1, the rapid but short-lived decrease of leaching followed by an increase and then a decrease in ammoniacal nitrogen leached is also prevalent in this case, with the good correlation of the leaching functions (Table 8.7) providing a very good result in the emission model (Figure 8.14).

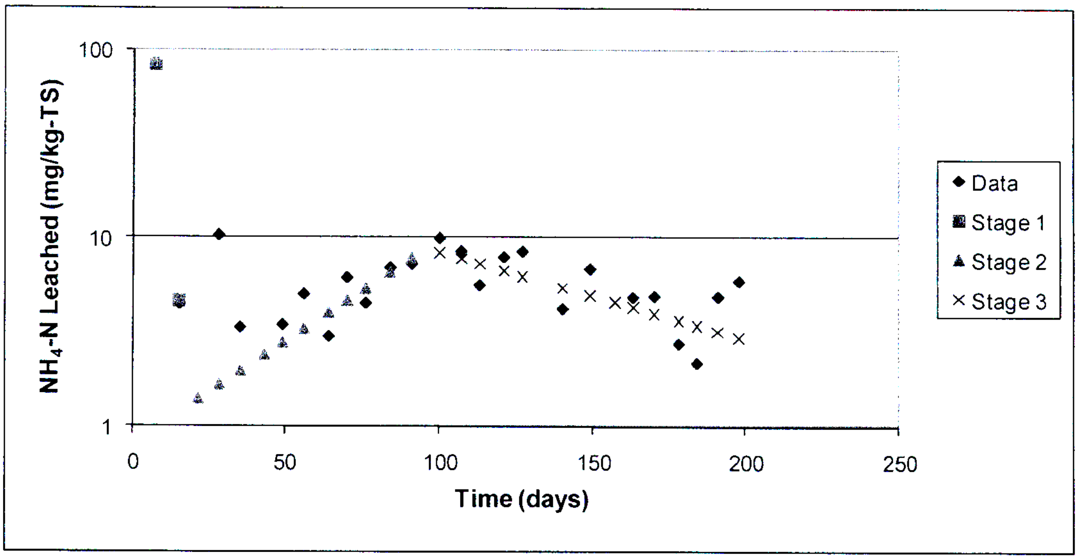

Figure 8.13 Column 2 NH_4-N Leaching Function

Table 8.7 Column 2 NH_4-N Leaching Function parameters and Emission Model Parameters

Stage	Lag (days)	a_j	b_j	Correlation
Ψ_{NH4-N} Stage 1	0	1057	-0.36	0.99
Ψ_{NH4-N} Stage 2	18	1.32	0.025	0.88
Ψ_{NH4-N} Stage 3	95	8.90	-0.011	0.67
Emission Model				1.0

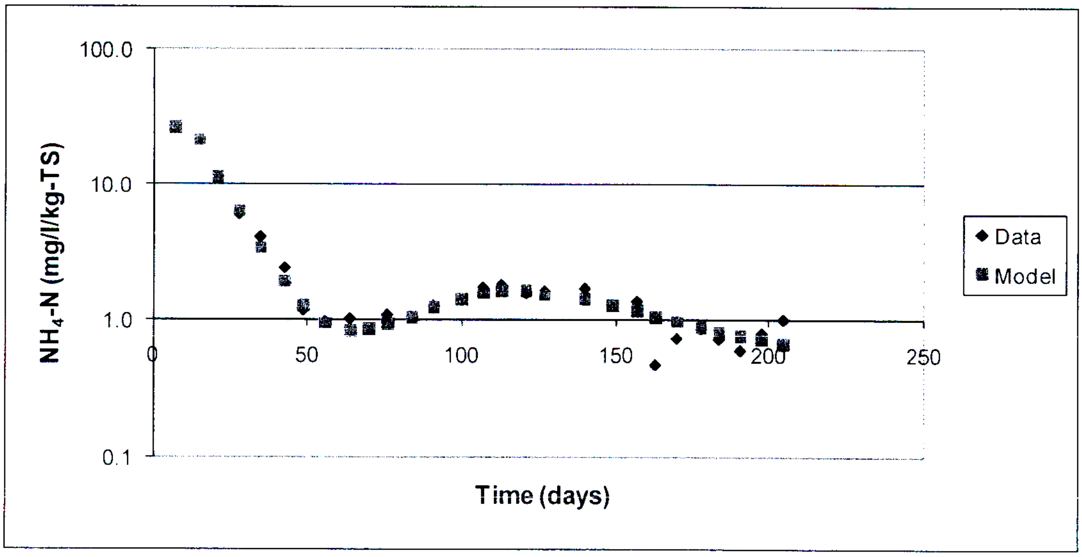

Figure 8.14 Column 2 NH_4-N Model and Data Comparison

Column 3 – 16 week coarse waste

The result from the assessment of the 16 week treated coarse material is similar to column 1 and 2 in terms of the evidence of three distinct stages of leaching (Figure 8.15).

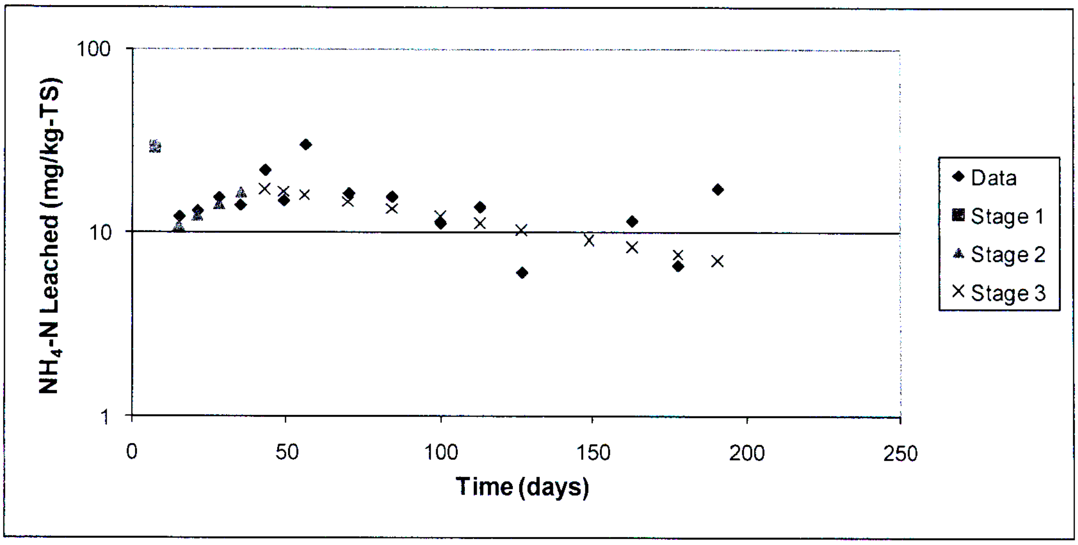

Figure 8.15 Column 3 NH_4-N Leaching Function

Table 8.8 Column 3 NH_4-N Leaching Function parameters and Emission Model Parameters

Stage	Lag (days)	a_i	b_i	Correlation
Stage 1	0	69.68	-0.12	na
Stage 2	15	10.82	0.021	0.80
Stage 3	38	17.8	-0.0061	0.35
Emission Model				0.99

The second stage shows a good correlation while the third stage correlation is weak (Table 8.8). Note that least squares regression was not used to fit the first stage as only two points were available, hence the correlation is not applicable. The emission model, however, provides a very good prediction of the actual data, apart from a slight underestimation between day 50 and 100 (Figure 8.16). This is due to the adjustment of the stage initiation in order to preserve continuity.

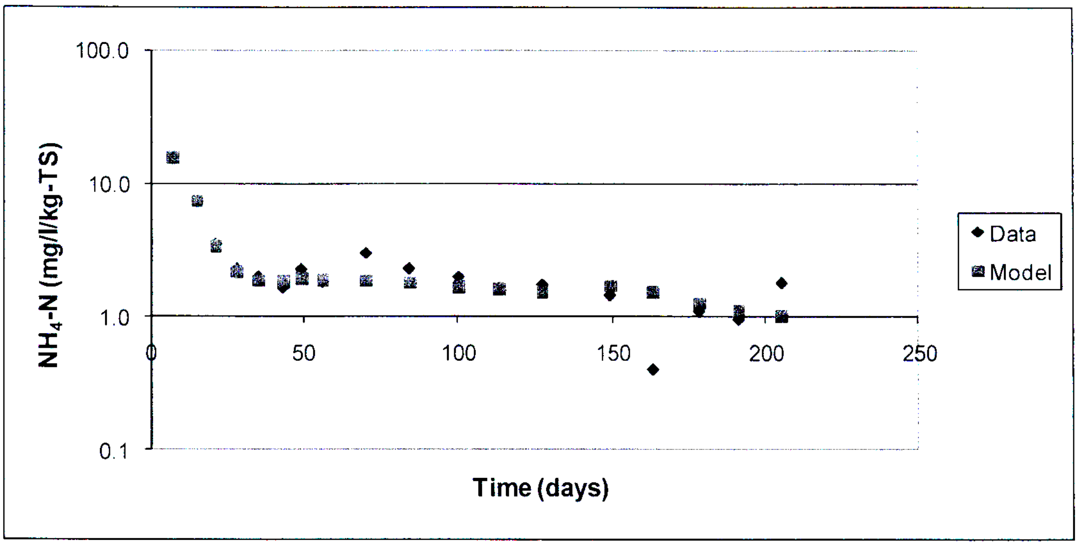

Figure 8.16 Column 3 NH_4-N Model and Data Comparison

Column 4 – 8 weeks treated fine waste

Only 1 stage was observed in the assessment of the leaching from the 8 week treated fine waste, as the rapid decrease and subsequent increase observed in the previous 3 columns are not clear in the data (Figure 8.17).

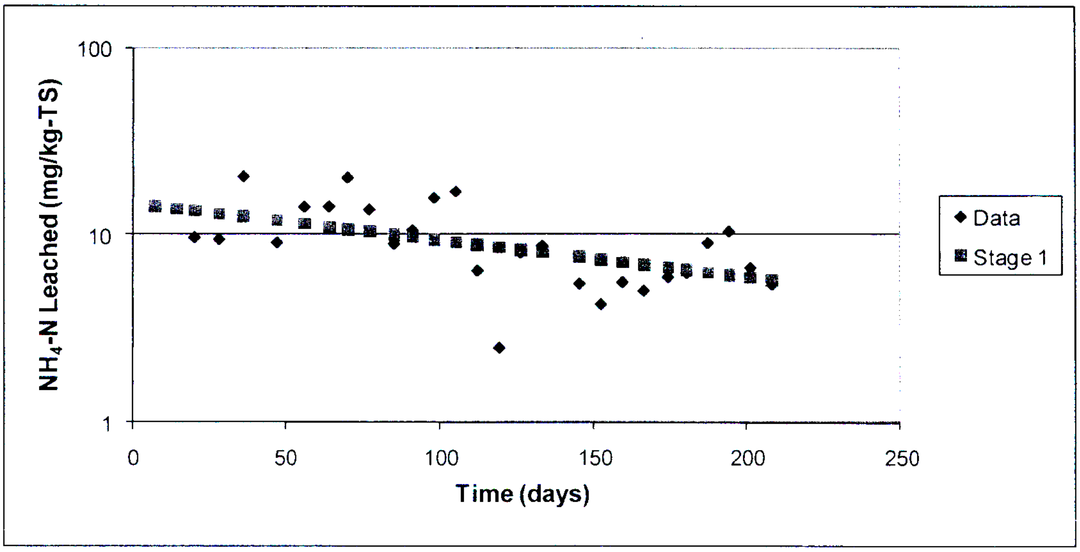

Figure 8.17 Column 4 NH_4-N Leaching Function

Table 8.9 Column 4 NH_4-N Leaching Function parameters and Emission Model Parameters

Stage	**Lag (days)**	**a_i**	**b_i**	**Correlation**
$\Psi_{NH4\text{-}N}$ Stage 1	0	15.00	-0.0046	0.57
Emission Model				0.84

The occurrence of only one stage may be due to the low levels of available nitrogenous compounds within the waste as a result of the treatment stage. The correlation of the leaching function is weak (Table 8.9) and the emission model does not provide very accurate estimations for the beginning of the test, but provides a reasonable fit to the actual data thereafter.

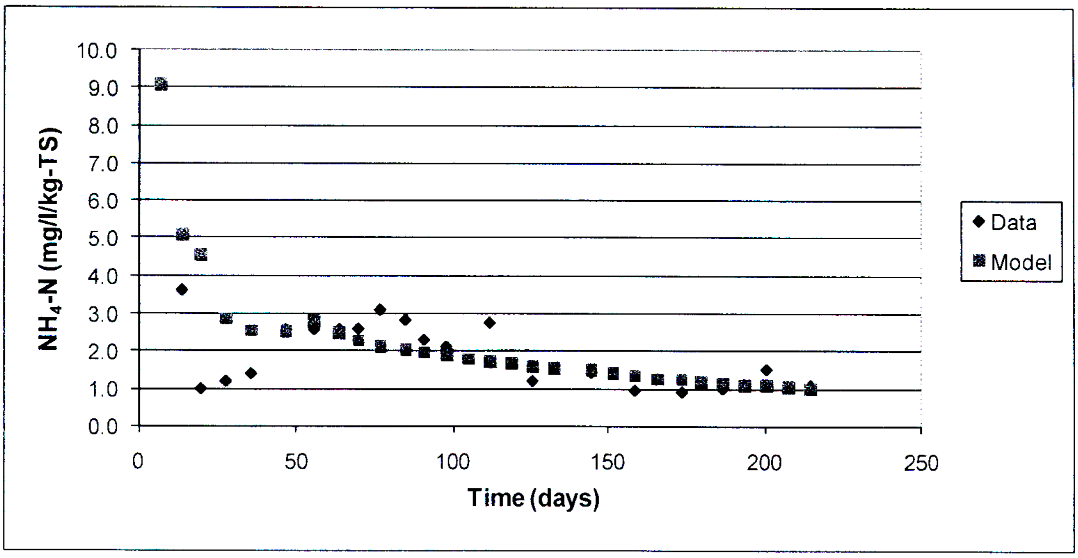

Figure 8.18 Column 4 NH_4-N Model and Data Comparison

Column 5 – 8 weeks treated coarse waste

The results of the fitting of the leaching function to the data from the 8 weeks treated coarse waste are presented in Figure 8.19.

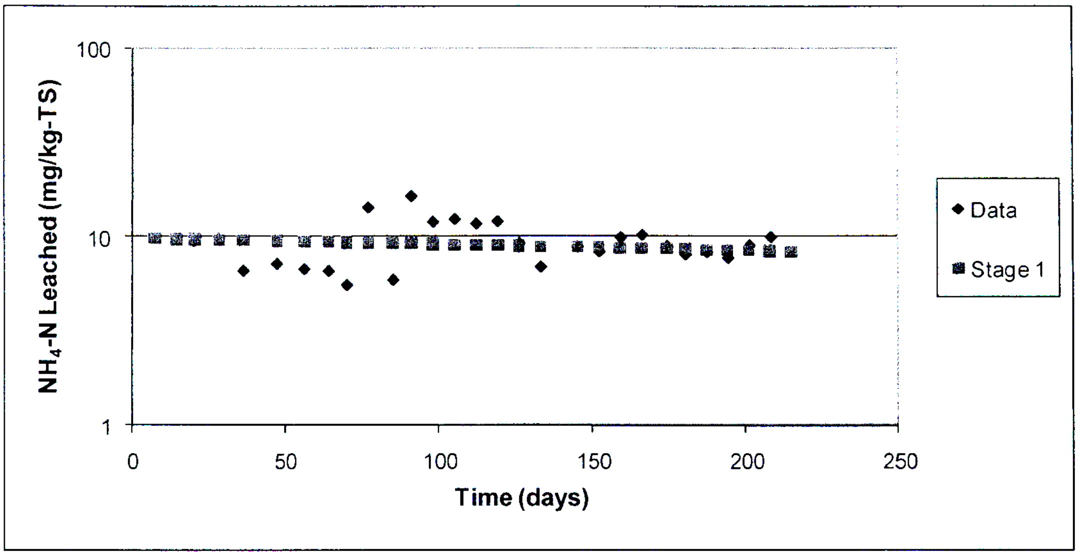

Figure 8.19 Column 5 NH_4-N Leaching Function

Table 8.10 Column 5 NH_4-N Leaching Function parameters and Emission Model Parameters

Stage	**Lag (days)**	a_i	b_i	**Correlation**
$\Psi_{NH4\text{-}N}$ Stage 1	0	10.0	-0.0079	0.51
Emission Model				0.98

As with the other 8 week treated waste column, only one clear stage is evident and a weak leaching function is formed from the data (Table 8.10). The emission model reflects the poor leaching function correlation with inaccuracies at the start of the test, however, as with Column 4, the emission model is more reliable from midway through the test with a very good and results in a very good correlation of 0.98 (Figure 8.20).

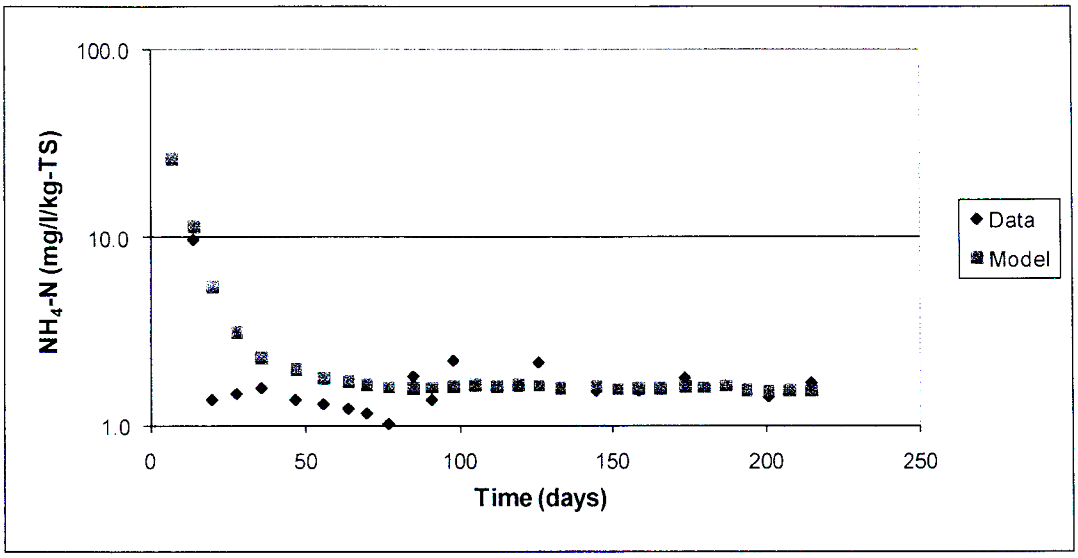

Figure 8.20 Column 5 NH_4-N Model and Data Comparison

Comments

Although a general pattern was not as distinct is in the case of the COD models, the first three column tests showed three stages of ammoniacal nitrogen leaching:

- Stage 1, the rapid decrease of the rate of solubilization would indicate the leaching out of the readily available ammoniacal nitrogen
- Stage 2 shows an increase in the rate of ammoniacal nitrogen solubilisation as the biological activity increases, with more proteinacious compounds being hydrolyzed by the bacteria
- Stage 3 reflects the decrease in ammoniacal nitrogen released as the quantity proteinacious compounds is reduced. This third stage is significant as a declining source term of ammoniacal nitrogen is difficult to determine (Environment Agency, 2003).

The results of Column 4 and 5 do not reflect the behavior of the other columns with only one stage evident in the results. This may be caused by a lower ammoniacal nitrogen release potential due to the more efficient DAT windrow treatment stage. Despite the differences in the solubilisation data, the model still provides a good prediction of the actual data.

The result of the column leaching study is significant as the combination of column leaching tests and modeling may allow for the long term prediction of ammoniacal nitrogen release from landfill sites.

8.4 LYSIMETER MODEL PARAMETERS

8.4.1 Imperfect Mixed Reactor Modeling

An additional uncertainty that is present in the lysimeters is the degree of mixing that occurs within the vessel. The presence of preferential flow paths results in a channeling of the input water which then bypasses a certain amount of the lysimeter material. Thus an imperfect mixed reactor liquid flow model is adopted. Due to the large degree of uncertainty in the pore structure of the waste material, determining the preferential flow paths and diffusive mixing would be a complex process and thus a simplification of the flow model is made. The diffusive mixing is considered to be slow in comparison to the preferential flow paths in the unsaturated conditions of the lysimeter and the diffusive mixing is thus ignored. The material involved in the leaching reactions is considered to be a fraction of the total lysimeter content and behave as a mixed reactor, thus the proportion of the lysimeter that is significantly affected by the liquid flow must be found.

The imperfectly mixed reactor assumption backed up by considering lysimeter moisture content, which remains well below the predicted field capacity value. If the active fraction of the lysimeter that is designated α , the leaching function (8.5) becomes:

$$\psi = \frac{C_{i+1} \cdot V_T \alpha}{M_{TS} \alpha} - \frac{C_i}{M_{TS} \alpha}\left(V_T \alpha - V_S\right) \qquad 8.10$$

To calculate the parameter (α), least squares regression on a multi stage exponential leaching model was applied to the range $0.1 \leq \alpha \leq 1.0$. The correlation of the functions was used as a measure of the best value for α . This value was then compared to ratio predicted by the expected field capacity. Both ammoniacal nitrogen and COD were used for this assessment and the results presented in Figure 8.21 and 8.22 for lysimeter 2 and 3 respectively. Lysimeter 1 COD was not modeled, as discussed below.

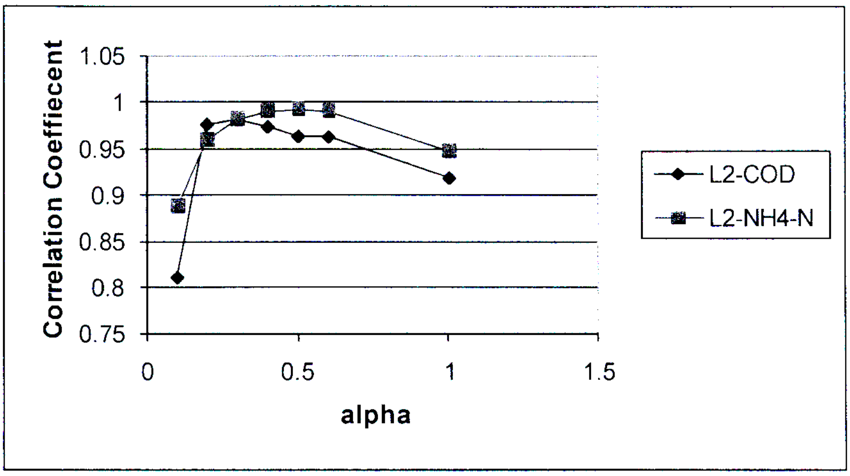

Figure 8.21 Correlation of Lysimeter 2 COD and NH_4-N for different α values

Lysimeter 2

From Figure 8.21 it can be seen that the best correlation is α = 0.3 and α = 0.5 for COD and ammoniacal nitrogen respectively. Thus a value of α = 0.4 is chosen for the final model calibration. This corresponds well to the lysimeter moisture content which remains approximately 40% of the expected field capacity throughout the test.

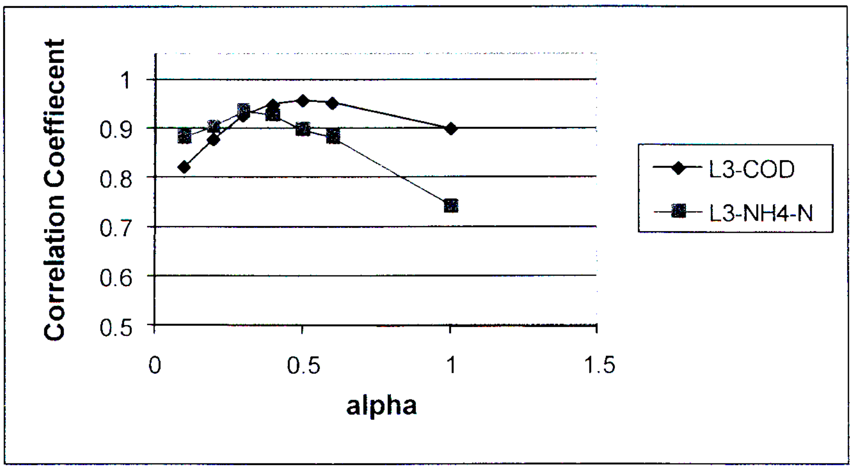

Figure 8.22 Correlation of Lysimeter 3 COD and NH_4-N for different α values

Lysimeter 3

Figure 8.22 shows that the best correlation is α = 0.5 and α =0.3 for COD and ammoniacal nitrogen respectively. Thus α = 0.4 is also chosen. This is not as well justified by the percentage of waste at field capacity which ranges from 50-70% during the test. This anomaly can be explained by the nature of the material, with large impervious fractions able to trap water and result in a higher field capacity than expected based on total waste mass.

Lysimeter 1

An assessment of the leaching of the untreated waste indicated that the consumption of compounds (negative leaching) is prevalent throughout the test and thus a leaching dominated removal model is not applicable. This is due to the high biological activity in the lysimeter and thus COD is not modeled. The results for the strength of the model correlation for the ammoniacal nitrogen model with different values for alpha are not conclusive as the correlation does not appear to be very sensitive to variations in the alpha values in the range of 0.4 to 1.0. (Figure 8.23). An alpha value close to 1.0 is highly unlikely and the proportion of waste at theoretical field capacity is 70%. As the material structure is expected to be a combination of the coarse and fine fractions as in Lysimeter 2 and 3, a value of 0.4 will be used.

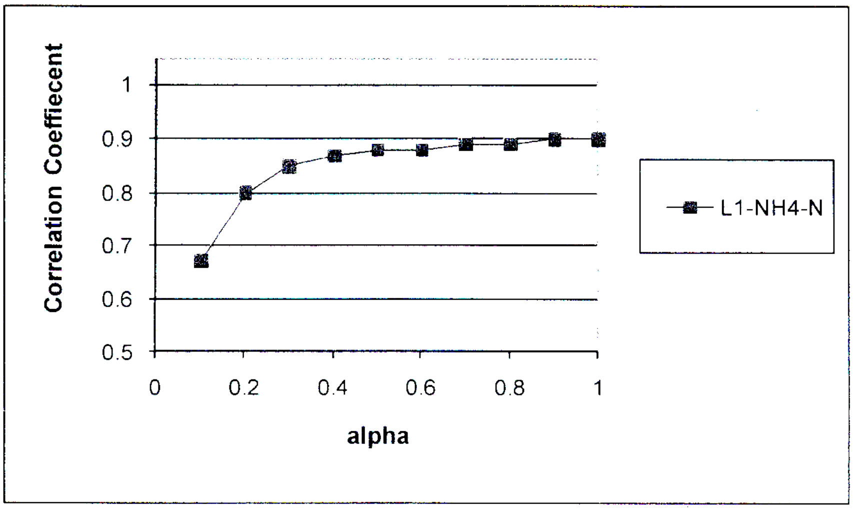

Figure 8.23 Correlation of Lysimeter 1 NH_4-N for different α values

8.4.2 Lysimeter COD Leaching Function and Emission Model

The establishment of the Lysimeter COD model follows a similar procedure to that of the columns, with the addition of the imperfect mixing parameter. A logarithmic scale is used to identify different leaching stages with correlation coefficients used to assess the models. As discussed, the COD for lysimeter 1 is not modeled.

Lysimeter 2 – 16 weeks treated fine waste

The assessment of the leaching of COD from the 16 week treated waste in Lysimeter 2 is presented in Figure 8.24.

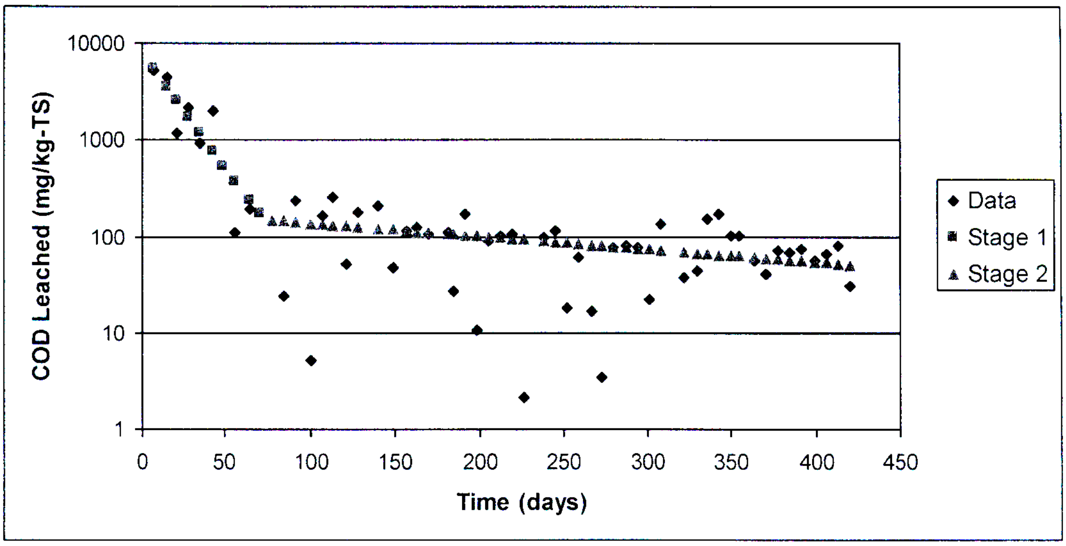

Figure 8.24 Lysimeter 2 COD Leaching Function

Table 8.11 Lysimeter 2 COD Leaching Function parameters and Emission Model Parameters

Stage	Lag (days)	a_j	b_j	Correlation
Ψ_{COD} Stage 1	0	7958	-0.055	0.92
Ψ_{COD} Stage 2	77	149.8	-0.0031	0.40
Emission Model				0.97

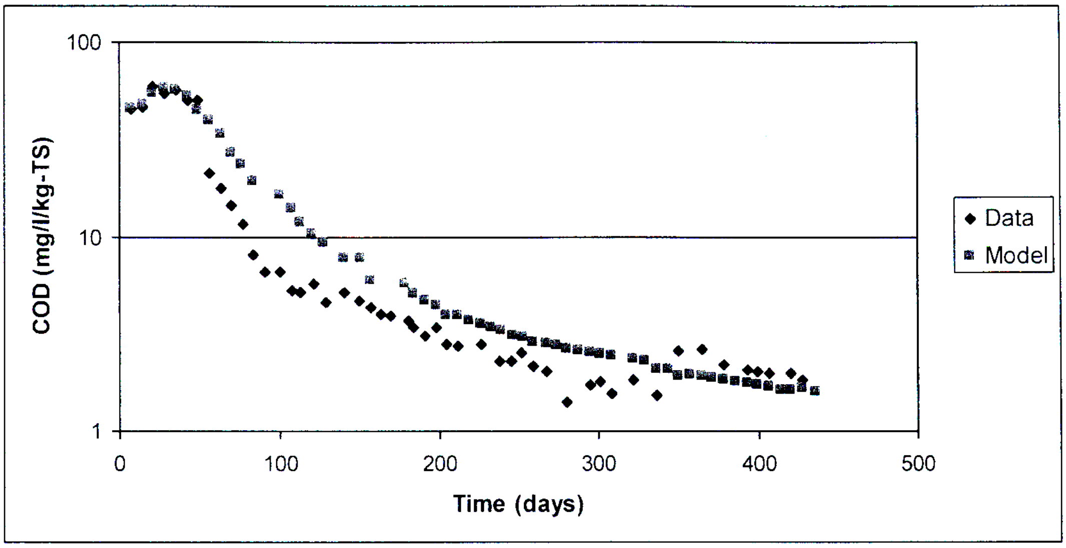

Figure 8.25 Lysimeter 2 COD Emission Model and Data comparison

Two stages in COD leaching are identified and functions are fitted (Figure 8.24). The first stage correlates very well although the second stage does not (Table 8.11). A more active biological component results in instances of net consumption (-Ψ) during the assessment and thus the final implementation overestimates the COD concentration. Despite this, a fairly reasonable result is obtained and the correlation coefficient for the Emission model is 0.97 (Figure 8.25). The progression from stage 1 to 2 corresponds roughly with the peak in methane concentration.

Lysimeter 3 – 16 weeks treated coarse waste

The leaching of lysimeter 3 containing the 16 weeks treated coarse fraction is plotted on a logarithmic scale (Figure 8.26)

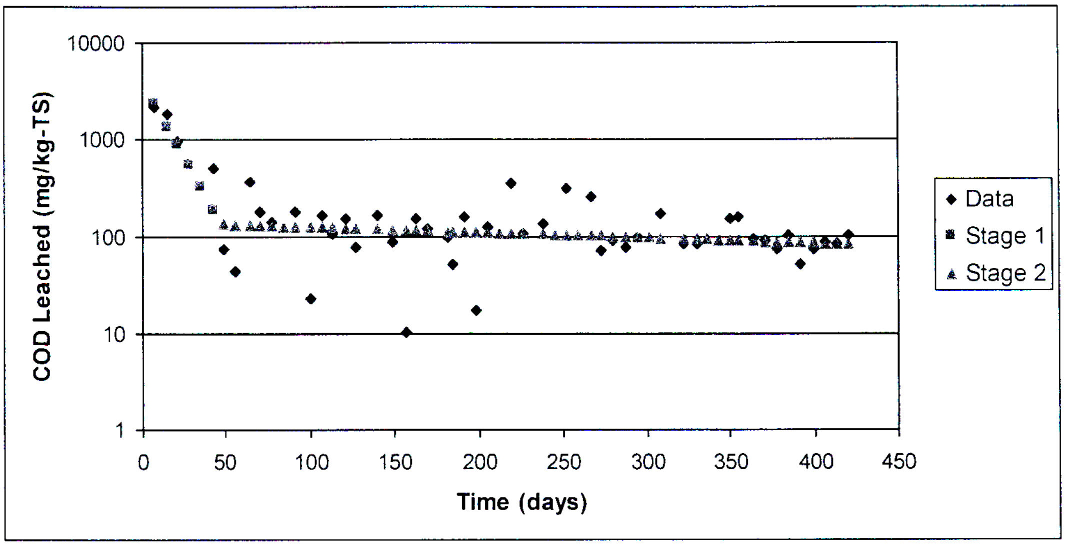

Figure 8.26 Lysimeter 3 COD Leaching Function

Table 8.12 Lysimeter 3 COD Leaching Function parameters and Emission Model Correlation

Stage	**Lag (days)**	a_j	b_j	**Correlation**
Ψ_{COD} Stage 1	0	3861	-0.07	0.96
Ψ_{COD} Stage 2	43	132.9	-0.0013	0.17
Emission Model				0.95

Two stages are identified in the analysis of Lysimeter 3 leaching data (Figure 8.26). The correlation for the first stage is very good but the second stage is very poor (Table 8.12). However, a good result is still obtained for the implementation of the emission model (Figure 8.27) with a correlation coefficient of 0.95. The start of the second stage coincides with the first measurement of methane within the lysimeter.

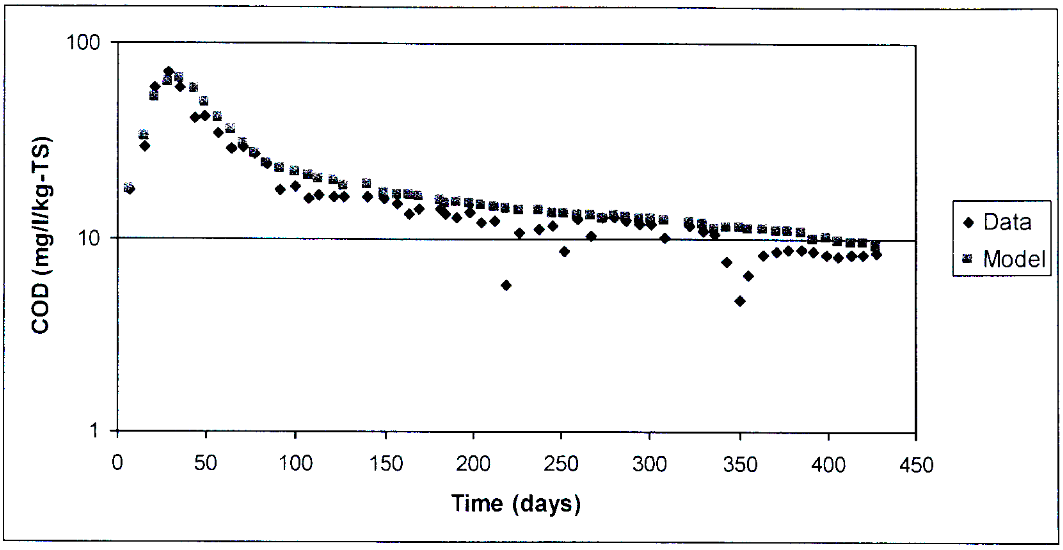

Figure 8.27 Lysimeter 3 COD Model and Data comparison

Comments

The implementation of the leaching only model does work for the treated waste, although neglecting the removal of compounds due to biological activity causes an overestimation of the results, particularly in the earlier stages of the test. The effect of this simplification is greater due to the smaller relative liquid flux and thus the consumption parameters have a greater influence on the results. In the long term, however, the model correlates well with the actual data.

8.4.3 Lysimeter Ammoniacal Nitrogen

As for the lysimeter COD, the release of ammoniacal nitrogen is modeled below for all three lysimeters.

Lysimeter 1- Untreated Waste

The logarithmic plot of the leaching of ammoniacal nitrogen from Lysimeter 1, which contains untreated waste, results in one clear stage (Figure 8.28). The leaching data shows a wide scatter and the correlation of the leaching function (0.27) is very weak.

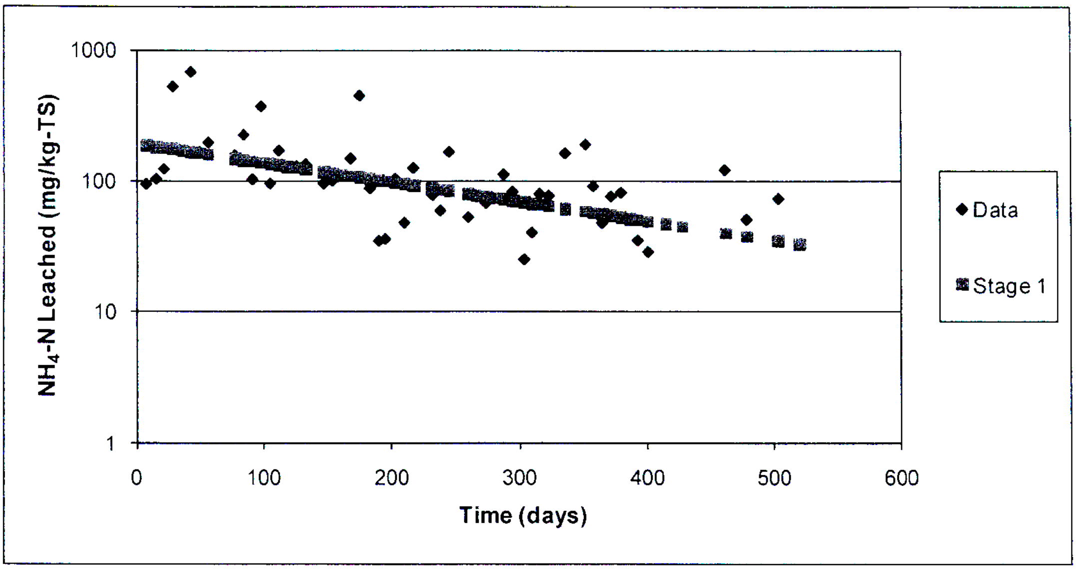

Figure 8.28 Lysimeter 1 NH_4-N Leaching Function

Table 8.13 Lysimeter 1 NH_4-N Leaching Function parameters and Emission Model Parameters

Stage	Lag (days)	a_i	b_i	Correlation
$\Psi_{NH4\text{-}N}$ Stage 1	0	196	-0.003	0.27
Emission Model				0.87

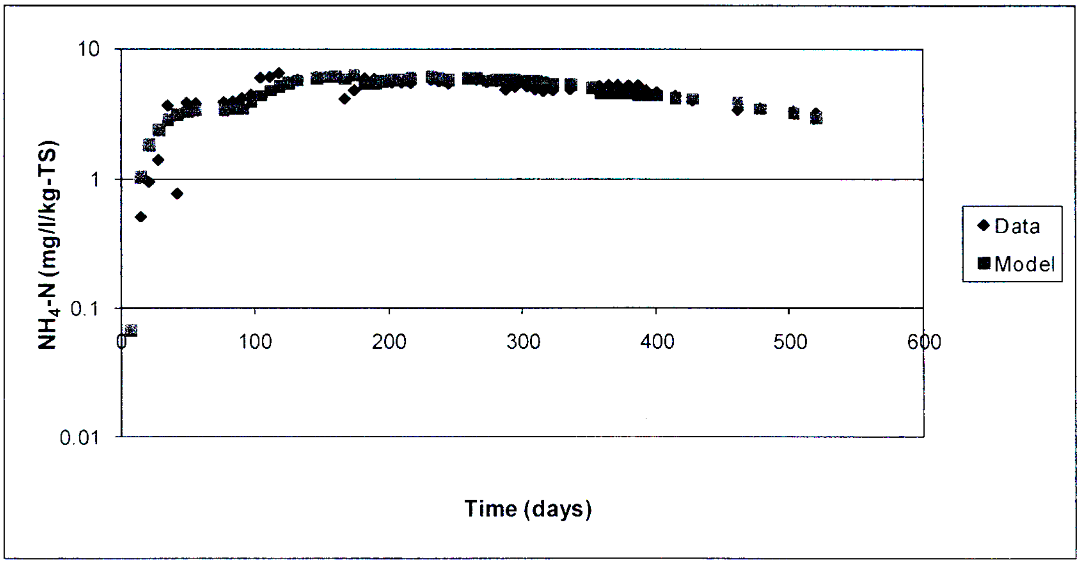

Figure 8.29 Lysimeter 1 NH_4-N Emission Model and Data comparison

Despite the poor correlation of the Stage 1 leaching function, the Emission model provides a good prediction of the data with a very good correlation coefficient of 0.87 (Figure 8.29)

Lysimeter 2 – 16 week treated fine waste

As with the column tests on the 16 week treated fine material, three stages are utilised to describe the ammoniacal nitrogen leaching (Figure 8.30). The correlation for the first stage is quite good (0.97), but it is weak for the second (0.48) and third stages (0.27) (Table 8.14). The short lived second stage of increasing ammoniacal nitrogen release occurs at the time of most significant methane generation. The Emission model predicts the beginning and end concentrations very well, but underestimates the data from day 100 to 200 (Figure 8.31). This is partly due to the establishment of continuity using the intersection method.

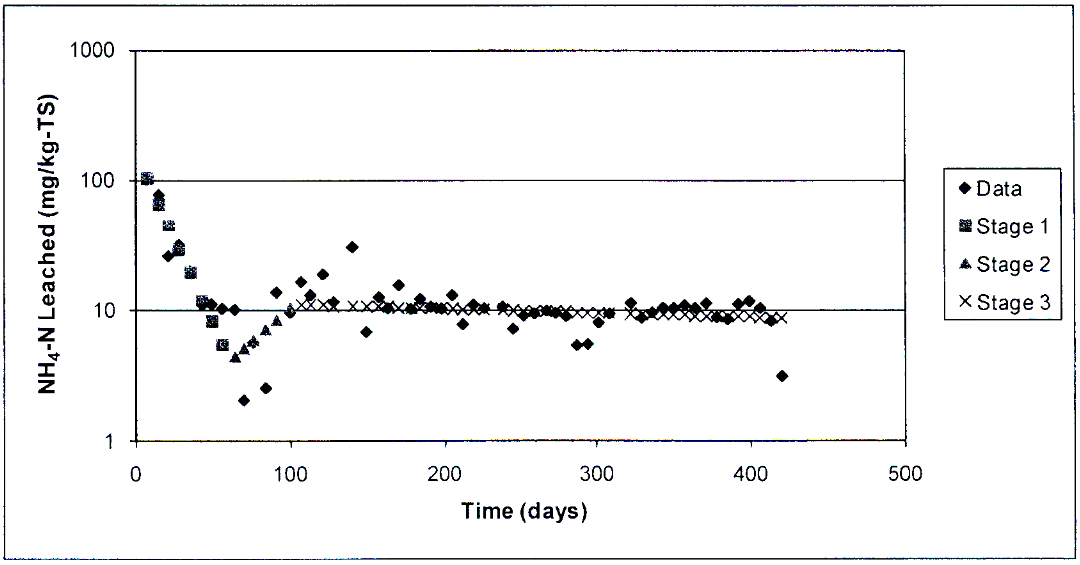

Figure 8.30 Lysimeter 2 NH_4-N Solubilisation Model

Table 8.14 Lysimeter 2 NH_4-N Leaching Function parameters and Emission Model Parameters

Stage	Lag (days)	a_i	b_i	Correlation
Ψ_{NH4-N} Stage 1	0	158	-0.06	0.97
Ψ_{NH4-N} Stage 2	43	2.65	0.024	0.48
Ψ_{NH4-N} Stage 3	107	10.9	-0.0007	0.27
Emission Model				0.99

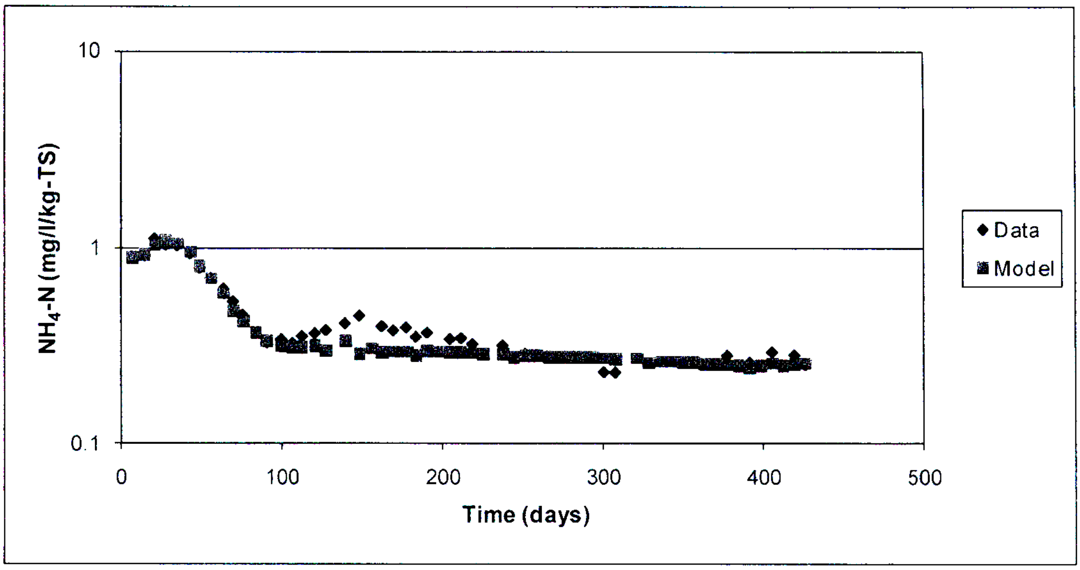

Figure 8.31 Lysimeter 2 NH_4-N Emission Model and Data comparison

The Emission model predicts the beginning and end concentrations very well, but underestimates the data from day 100 -200 (Figure 8.31) and when inspecting the leaching curves it is apparent that this is largely due to the establishment of continuity using the intersection method. The correlation coefficient of 0.99 the Emission model is very strong.

Lysimeter 3

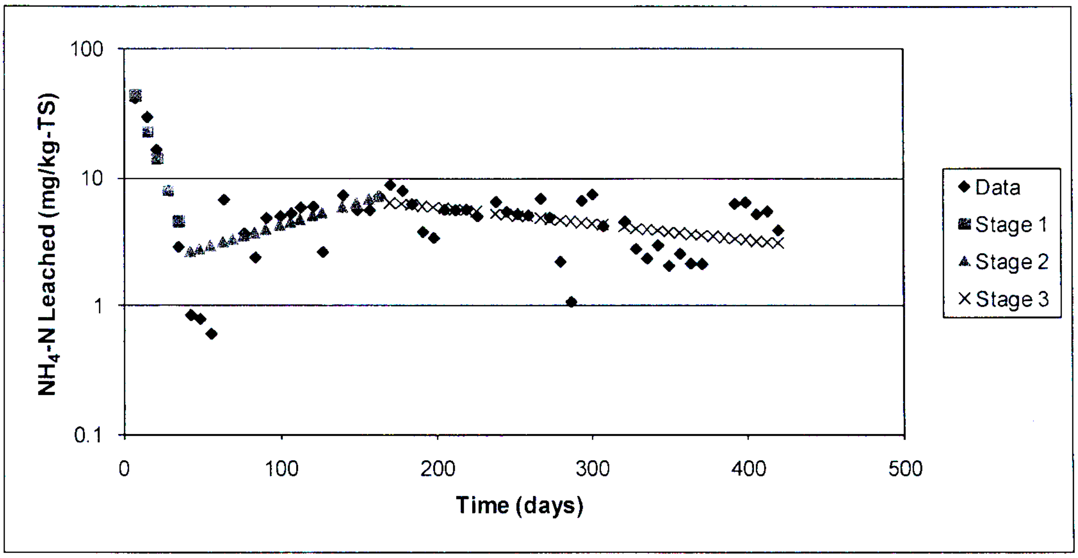

Figure 8.32 Lysimeter 3 NH_4-N Leaching Function

Table 8.15 Lysimeter 3 NH_4-N Leaching Function parameters and Emission Model Parameters

Stage	Lag (days)	a_i	b_i	Correlation
Ψ_{NH4-N} Stage 1	0	79.7	-0.081	0.98
Ψ_{NH4-N} Stage 2	43	2.64	0.0084	0.42
Ψ_{NH4-N} Stage 3	170	6.42	0.003	0.41

Three stages are also used in the leaching of Lysimeter 3 (Figure 8.32), with second and third stages showing a weaker correlation than the first stage (Table 8.15). The model predicts the ammoniacal nitrogen very well, but overestimates the readings from day 60 to 120 (Figure 8.33). This may also be attributed to errors associated with the intercession method.

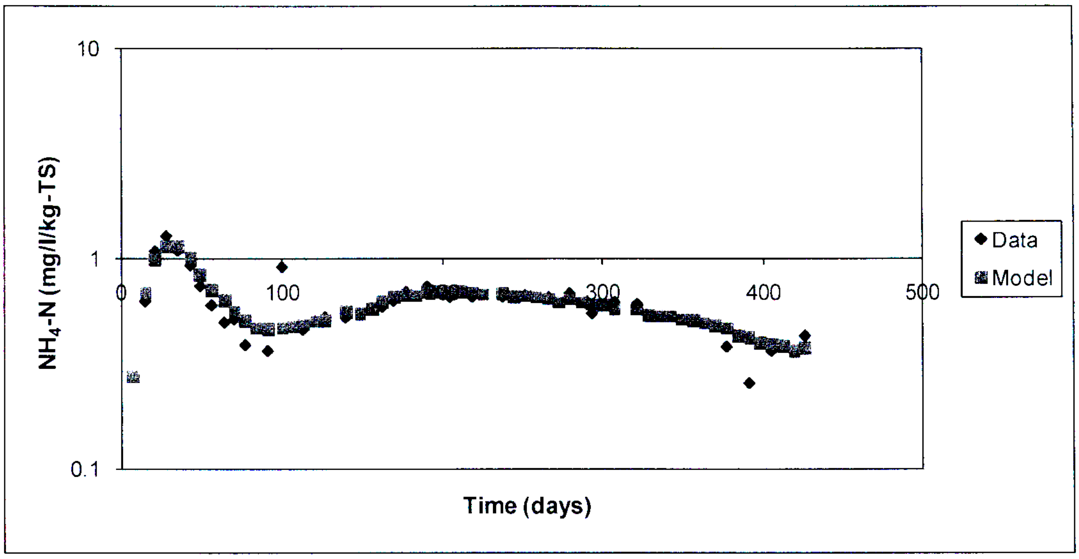

Figure 8.33 Lysimeter 3 NH_4-N Model and Emission Data comparison

Comments

The 3 stage leaching of ammoniacal nitrogen seen in the column study was also observed in Lysimeters 2 and 3 is a good means of characterising the progression of ammoniacal nitrogen release from waste. Lysimeter 1 did not display multiple stages and this may be because the system had not yet entered the fully methanogenic stage. Although inaccuracies are encountered from the establishment of continuity, the Emission Model for ammoniacal nitrogen release is more reliable than the COD emission model. This is expected due to the absence of ammoniacal nitrogen consumption in the anaerobic conditions and thus the disregarding of biological removal of the ammoniacal nitrogen is a reliable assumption.

8.4.4 Comparison of Leaching Functions

The leaching functions are summarised in figures 8.34 and 8.35 which provide a means of direct comparison between the different wastes tested.

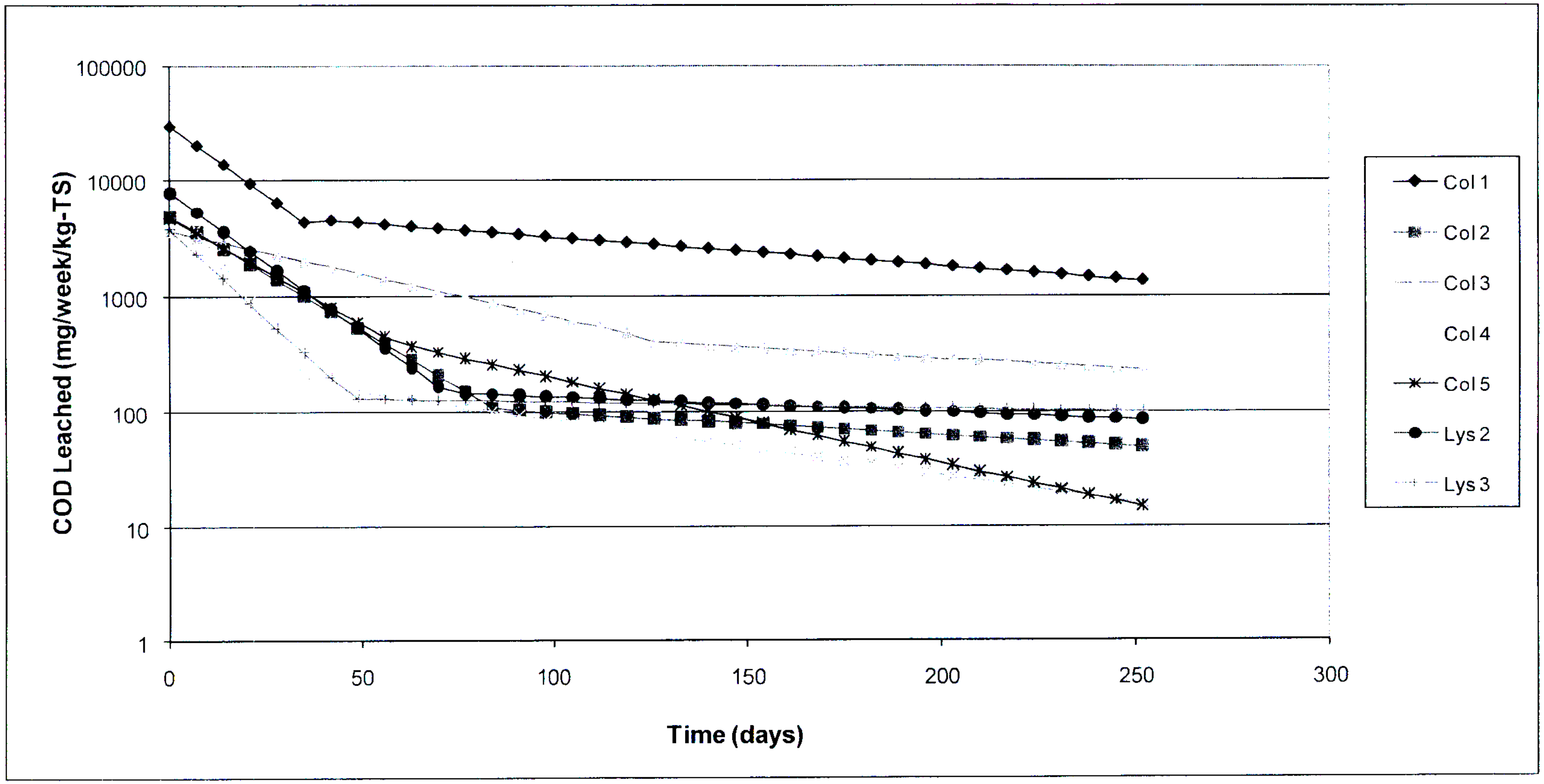

Figure 8.34 Comparison of COD Leaching Functions

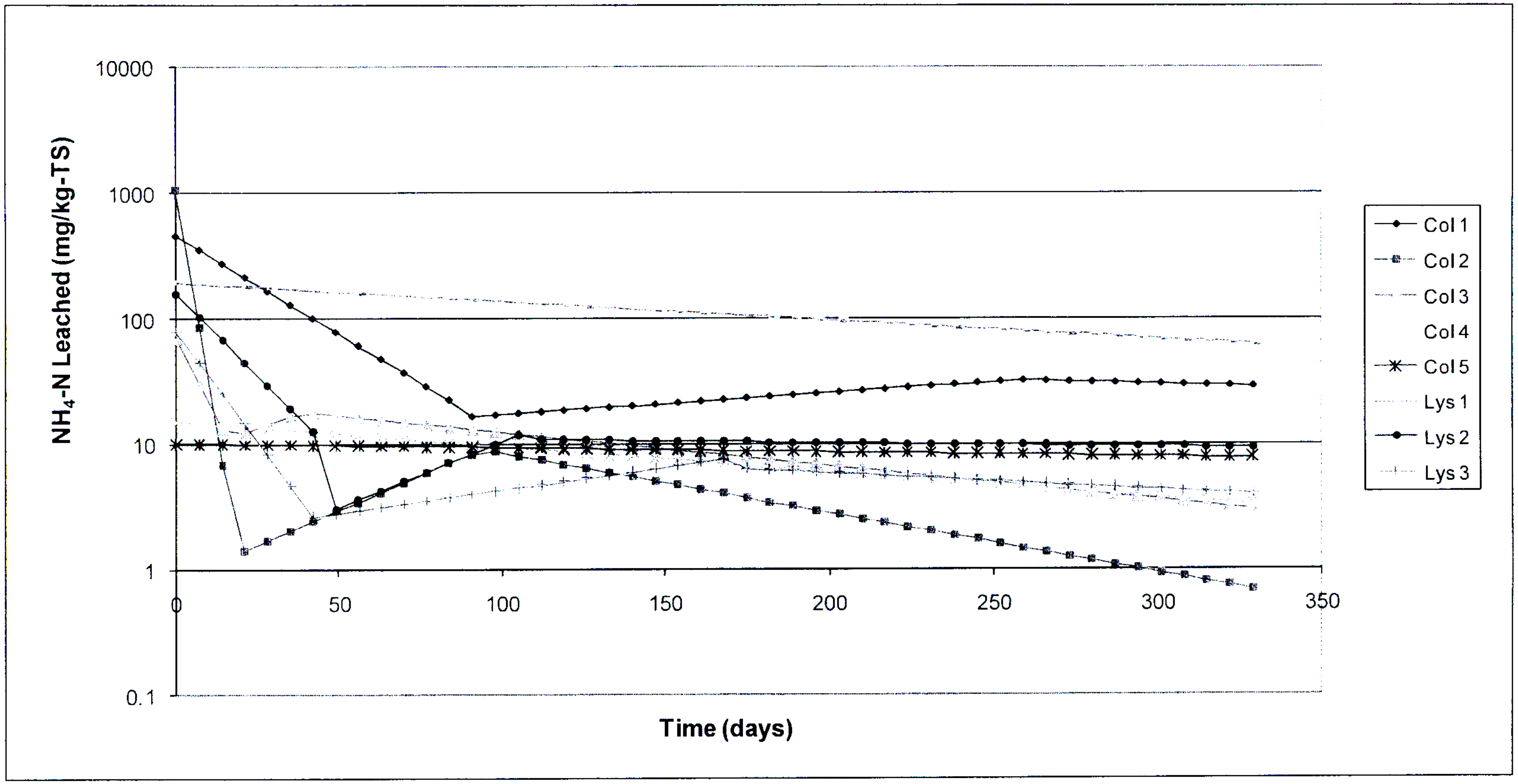

Figure 8.35 Comparison of NH_4-N Leaching Functions

COD

By comparing the COD results, it is clear that Column 1 has by far the greatest leaching potential. The treated waste samples have fairly similar leaching trends, with the initial leaching (given by parameter a) falling into the range of 3200 to 3900 mg/kg/week for the treated waste columns, and 2300 to 5400 mg/kg/week for the lysimeters. The rate of reduction in leaching is fairly similar for all the tests conducted, particularly the second stage decay. Of particular interest is the very close correlation between lysimeter 2 and 3 second stage, as well as the similarity between lysimeter 2 and column 2 which contain the same material. There are similarities between lysimeter 3 and column 3 in the initial leaching value, as well as the slope of the second stage decay curve. Column 3, however has a slower first stage decay. This could be attributed to the delay in methanogenic activity within the column due to oxygen inhibition (Figure 7.21).

Ammoniacal nitrogen

The comparison of the ammoniacal nitrogen leaching curves (Figure 8.35) shows a lesser degree of similarity between the 7 tests. The untreated waste tests – Lysimeter 1 and Column 1 remains the highest in terms of leaching potential, and has a protracted second stage of increasing ammoniacal nitrogen release. Column 3 and Lysimeter 3 are very close in this case, however, Column 2 and Lysimeter 2 do not correlate well, although the second growth stages are very close. Lysimeter 1, containing untreated waste, has the greatest initial rate of leaching which decreases at a far lower rate than that of Lysimeter 2 and Lysimeter 3 which contained treated waste.

8.5 ASSESMENT OF RESULTS USING LEACHING MODELS

The emission models make it possible to draw comparisons between the different waste types tested.

8.5.1 Total COD and Ammoniacal Nitrogen Release

The leaching models developed will be used to extrapolate the results in order to predict the ultimate cumulative release of pollutants from the waste in the form of COD and ammoniacal nitrogen. Where the model overestimates or underestimates the actual figure given by the test data, the figure is adjusted before the extrapolation. In the case of Lysimeter 1 COD, where the model could not be applied, the leaching from Column 1 was used in the extrapolation. The results of the cumulative release are presented in Table 8.16 and compared to the cumulative load calculated from the eluate tests.

Table 8.16 Total COD and NH_4-N release from Eluate tests and Model Predictions

Waste Type	Units	Cumulative COD		Cumulative NH_4-N	
Untreated		(End Test)	Projected	(End Test)	Projected
Eluate		75987		485	
Column 1		(158254)	170600	(2097)	2140
Lysimeter 1		(77507)	80500	(2023)	2738
16 Weeks Fine					
Eluate		34230		11.1	
Column 2		(26933)	28940	(1002)	1090
Lysimeter 2		(9211)	10220	(281)	965
16 Weeks Coarse	**mg/kg-TS**				
Eluate		34570		9.5	
Column 3		(34481)	43930	(428)	590
Lysimeter 3		(10012)	19950	(306)	475
8 Weeks Fine					
Eluate		12430		14.6	
Column 4		(13936)	14304	(338)	520
8 Weeks Coarse					
Eluate		17390		10.4	
Column 5		(29563)	29665	(485)	1911

The final estimated release for all the tests shows that the untreated waste carries the greatest loading by a large margin. When comparing treatment time within the same material size, the 16 weeks treated waste has a greater COD release than the 8 weeks waste, although the ammoniacal nitrogen values are lower. The 16 weeks material, coarse and fine, return similar values for both COD and ammoniacal nitrogen, but in the case of the 8 weeks treated waste, the fine material has a significantly lower cumulative release than the coarse material based on the actual recorded data, with an even greater difference in the extrapolated results. This result however, may be exaggerated due to inaccuracies in the leaching function.

In terms of comparisons between the three different types of testing, the eluate tests are similar to the results of the column tests for both 16 weeks waste types and for the 8 weeks fines, and lower than the results obtained for the untreated waste and 8 weeks coarse waste. The columns return a higher value than the lysimeters for COD released, as expected due to the consumption of compounds through microbial activity in the lysimeters. As expected, the ammoniacal nitrogen values were higher in the extended anaerobic tests of the lysimeters and columns than for the eluate tests. Thus the eluate tests do provide an insight in terms of the COD of a material, but are unreliable for the assessment of ammoniacal nitrogen.

8.5.2 Long Term Emission at Full Scale

The emission models formulated from the 15 different tests are applied to a theoretical full-scale landfill scenario. A landfill, 20m deep with a 500kg/m^3 dry density and a field capacity of 55%. As used by Leikam et al, 250mm/m^2 will be used as an infiltration rate. The liquid model is based on 40% mixing with the infiltration spread over weekly intervals.

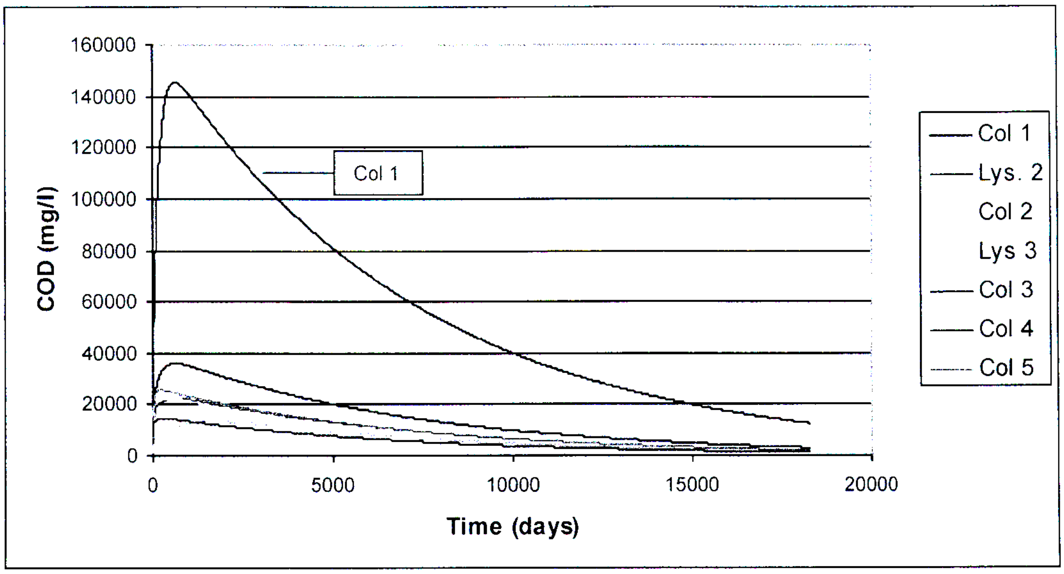

Figure 8.36 Projected full scale COD concentrations

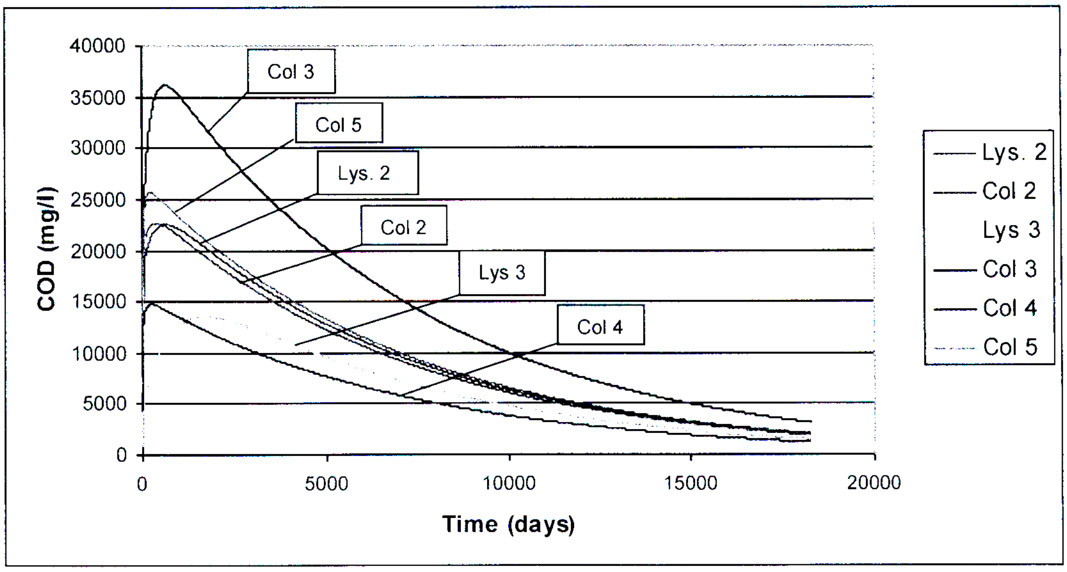

Figure 8.37 Projected full scale COD concentrations excluding Column 1

Figures 8.36 and 8.37 show the predicted concentration of COD over a 50 year period. Note that the result for Column 1 is not presented in Figure 8.37 in order to improve the readability of the other results. The benefit of waste treatment in terms of COD leachate loading is reinforced by this calculation, with the untreated waste showing a far greater COD load. When considering the results of the treated waste some interesting points should be highlighted. The similarity between column 2 and Lysimeter 2, tests containing the same material, is very encouraging when considering the large scale differences. Column 3 and Lysimeter 3 show a poor correlation, although, as mentioned previously, there were problems in the establishment of methanogenic activity in Column 3. When considering the difference between coarse and fine material, the fine columns (2 and 4) are significantly lower than the coarse columns (3 & 5), although Lysimeter 3, containing coarse material is lower than Lysimeter 2 (fine). The final point to note is that despite high variations in the peak concentrations of the various materials, the tail concentrations are all fairly similar. This has two implications: firstly, the sensitivity of the tail concentrations to the solubilisation reactions is low and secondly, the degree of waste treatment may not have a significant effect on the long term emissions of the landfill. Thus a shorter treatment timeframe could be adopted if the long term emissions are of the greatest concern to the waste managers. Finally it should also be noted that although the prediction of COD release allows for the comparison of the different waste materials in the context of these tests, the COD concentrations estimated at the full scale are too high due to the exclusion of the biological degradation reactions. Thus future work on this model must include an estimation of the biological consumption.

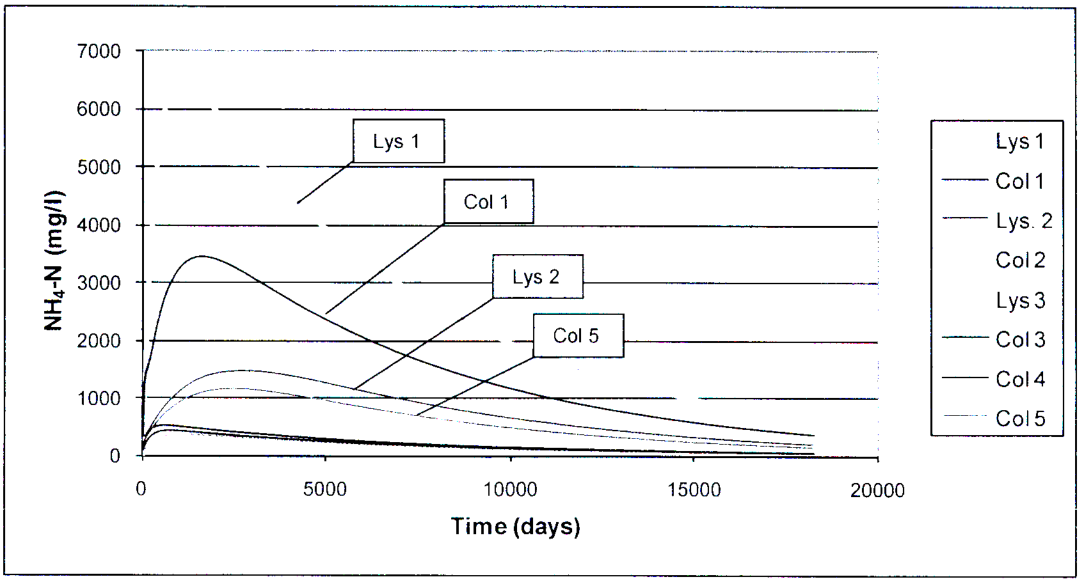

Figure 8.38 Projected full scale NH_4-N concentrations

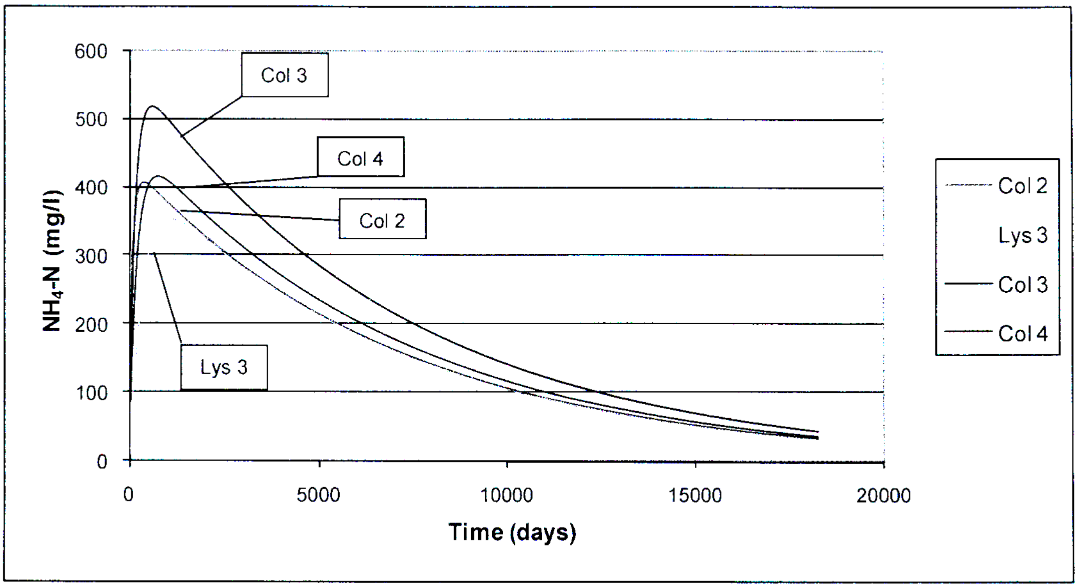

Figure 8.39 Projected full scale NH_4-N concentrations excluding Column 1, Lysimeter 1, Lysimeter 2 and Column 5

The ammoniacal nitrogen emissions from Lysimeter 1 and Column 1 (Figure 8.38) show the high ammoniacal nitrogen concentrations that can be expected from untreated waste, with lysimeter 2 and column 5 also showing a significantly higher ammoniacal nitrogen concentration than the other treated waste tests and may be due to inaccuracies in the leaching function. It should be noted that due to the incomplete establishment of the methanogenic stage in lysimeter 1, the results from the model could be too high. Of the other tests (Figure 8.39), the results are fairly. The ultimate values obtained should be more reliable than the COD values as there is no anaerobic consumption of ammoniacal nitrogen and thus the model is closer to the actual reactions. The difference in coarse and fine waste does not appear to be as distinct as that of the COD.

Discharge Limits

The discharge limit of 15mg/l-N for ammoniacal nitrogen is not achieved by any of the treatment methods within the 40 year full-scale prediction (Table 8.17). Thus, the best estimate at this stage is that landfill aftercare and leachate treatment beyond 40 years will still be required for landfills which have received treated waste. However, the timeframe could be reduced if the infiltration rate were increased. For example, Column

4 achieves the discharge limits within 40 years if the infiltration rate were increased from 250mm/m^2/annum to 425mm/m^2/annum. Considering the average annual rainfall for the region is 1009mm, this may be a plausible option, although it has obvious implications on the sizing of the leachate treatment works.

Table 8.17 Projected Results from Full Scale Scenario

Test	**Units**	**COD**		**NH_4-N**	
		Peak	40 yrs	Peak	40 yrs
Column 1	mg/l	145522	12577	3442	616
Lysimeter 1				6515	1027
Column 2		22651	1908	409	56.3
Lysimeter 2		22572	1997	1484	350
Column 3		36152	3147	519	55.9
Lysimeter 3		13537	1497	355	74.6
Column 4		14769	1197	416	61.5
Column 5		25787	2070	1175	261

8.5.3 Discussion of Model

The results obtained from the implementation of the model show that the emission model does have applications. The similarities in the first order decay constants (b_j) in materials of the same origin as well as the close result between Column 2 and Lysimeter 2 COD predictions at a simulated full scale show that the use of column leaching tests in conjunction with the model can deliver repeatable results. However, with only one test conducted for each material type, as well as anomalies, such as the problems with Column 3 and the high ammoniacal nitrogen concentrations of Lysimeter 2 and column 5, the application of the model parameters is limited to these specific tests. Thus further research should be conducted with the specific aim of establishing the leaching function and biological consumption of the COD, with multiple tests conducted on the same material type in order to improve the reliability and repeatability of the results.

8.6 SUMMARY OF MATERIAL TESTS

Of the extensive array of material tests, pollutant modeling and discussions presented in this results chapter, a common thread emerges. Without question, the benefit of treating waste is clear with the highest pollutant loading attributed to the untreated waste in all cases. The peak COD concentrations can be expected to be lower in treated wastes due to the avoidance of the high strength acid phase leachates, although in the long term, the effect of biological consumption of the organics would imply that the concentrations of COD from mature untreated landfill leachate is likely to be similar to that from the treated waste. The comparison between the 8 week and 16 week waste treatment is of no benefit due to the inefficiencies in the 16 week DAT process. The comparison between the fine and coarse material showed that firstly, based on physical characteristics, the coarse material is predominantly paper and plastic. The fine material carries a lower COD loading in all cases but in the case of the 16 week treated waste, the ammoniacal nitrogen loading was higher. This was not the case in the 8 week waste tests. The predicted release based on the full-scale scenario show that the coarse material has more leachable ammoniacal nitrogen.

Despite the numerous questions regarding the reliability of the tests, the following conclusions are made:

- The treatment of waste reduces the emission potential of waste in terms of both peak COD and ammoniacal nitrogen loadings and long term ammoniacal nitrogen emissions.
- The fine fraction may carry a smaller pollutant loading than the coarse fraction, although in the projected results the difference is less obvious.

The effect of treatment time could not be properly answered due to the suspected inefficiency of the 16 week DAT process. However, it can be concluded that the treatment of waste for 8 weeks is of significant benefit, and, based on literature, the effective treatment of waste for a longer timeframe should provide a material more stable than that of the waste 8 week treatment conducted in this study.

CHAPTER 9

APPLICATION OF FINDINGS TO FULL SCALE OPERATIONS

9.1 WASTE TREATMENT USING THE DAT WINDROW SYSTEM

The findings of this research are used for recommendations of a full scale waste disposal operation. The recommendations broadly cover the aspects of a waste treatment operation and costs and the landfill site operation and closure plan. The composting of municipal waste in Windrow 3 using the DAT proved to be successful and may be pursued at full scale. The conservative sizing of the windrows proved to be an unnecessary precaution, with ample oxygen available to the majority of the waste.

9.2 WINDROW SIZE AND WASTE RATIO

The high levels of oxygen and low occurrence of methane in the DAT windrow tests indicate that the conservative approach adopted in terms of windrow size and waste to structural material ratio was not necessary. Thus the same windrow design as that used in Cottbus should be used. The design parameters are as follows:

- Windrow base: 11m including cover
- Windrow height: 3m including cover
- Waste Ratio: 2 parts MSW to 1 part structural material

This design will provide a more efficient operation in terms of waste treated per dome and channel used and also in terms of the area requirement. Furthermore, less structural material, which is likely to be a limiting factor in the construction operation, will be required per unit volume of MSW treated. After the problem of the DAT windrows drying out too early, irrigation of the windrows, should be investigated further.

9.3 COMBINATION OF MRF WITH DAT SYSTEM

The South African waste stream is not separated at source and thus contains a large fraction of recyclable materials. By combining the process with a materials recycling facility, the quantity of inert material in the process will be reduced, thus improving the efficiency of the process. Furthermore, the MRF operation includes the breaking open of refuse bags which, if left sealed, would form anaerobic zones within the windrow. The use of a MRF would negate the need to shred or crush the majority of the MSW, however shredding of the structural material will still be required. Sieving the material after the treatment process will allow for the recovery of the scarce structural material as well as residual recyclables. The coarse material in general contains material of high calorific value and may be a valuable fuel source if a waste to energy plant is commissioned.

9.4 TREATMENT OF GARDEN REFUSE

A problem that was apparent is the limited supply of garden refuse for utilisation as the structural component of the windrow material (11 times less than MSW). The shortage is compounded by the fact that not all of the garden refuse, i.e. the "green, leafy" fraction, is suitable for use as the structural component.

A garden refuse composting operation for the "green and leafy" garden waste should be implemented, where the product is not landfilled, but redistributed for agricultural or horticultural usage. This is beneficial in terms of sustainable use of the waste and the saving of landfill airspace. The structural material from the garden refuse should then be used for the MSW MBT operations. As stated above recycling of the structural component of the windrowed material after the composting step may provide enough structure for the treatment requirements.

9.5 DAT OPERATION SPACE REQUIREMENTS

The DAT composting process has specific space requirements and the space availability is a limiting factor in terms of the composting process, as the energy input is irrespective of the degradation time. If space is at a premium, a shorter composting period would be adopted, although this should not fall below 8 weeks without further research into shorter composting timeframes. If there is still not enough space for the system, the option of treating the waste at alternative locations such as waste transfer stations should be considered. If however, additional space is available it would allow for the implementation of a longer aerobic treatment, depending on windrow moisture levels.

In order to implement the DAT at Bisasar Road the space required must be evaluated. Bisasar Road is the busiest landfill in South Africa and if implemented at full scale it would represent the largest MBT operation in the country with the waste quantities currently landfilled at the Bisasar Road Landfill Site (Courtesy of DSW) is as follows:

MSW	468 805t
Garden Refuse	40 922t
Mixed Loads	12 653t
Builder's Rubble	93 466t
Cover Material	316 897t
Other	44 186t
Total	976 911t

Table 9.1 Space Requirement for DAT operation at Bisasar Road

Parameter	**Unit**	**Value**	
Waste Requiring Treatment			
MSW:	t	468 000	
MSW Density (Bulk):	kg/m^3	300	
MSW Volume:	m^3	1 560 000	
Structural Material/MSW ratio	-	2:1	
Structural Material Required	m^3	780 000	
Structural Material Density	kg/m^3	200	
SM Required		156 000	
Total Mass	t	624 000	
Windrow Density	t/m^3	0,52	
Windrow Volume	m^3	1 200 000	
Treatment Period	weeks	8	20
Volume per MBT cycle	m^3	185 000	460 000
Windrow Unit volume	m^3/m	21,25	
Windrow Length	m	8 700	21 600
Space req per unit length*	m	16	
Total Area Requirement	**ha**	**13.9**	**34.6**

*Based on 11m windrow width + 5m for machinery access

The total quantity requiring treatment is comprised of garden refuse and MSW, which equates to approximately 510 000t per annum. Using the recommended windrow characteristics presented above the space requirement is calculated and presented in table 9.1.

The total footprint area of the Bisasar Road site is 44 ha, however, not all the space is suitable due to steep gradients. Thus implementation of a full-scale operation with an 8-week treatment period may be possible completely on site with longer treatment times requiring alternative locations for additional DAT operations.

9.6 LANDFILLING AND LEACHATE MANAGEMENT

The issue of landfilling operations is intrinsically linked to the leachate generation of the site, as the high compaction rates that are reported in the literature. This, however, would reduce the flux of liquid through the waste and thus the timeframe for stabilisation of the material would increase. However, an important consideration in the closure of the site is dealing with the large volumes of leachate that may be generated if a high flux of liquid is encouraged. It is thus recommended that high compaction rates are used and the emphasis placed on a robust leachate treatment system to deal with the latent levels of pollutants. Although this may be thought to be a process of dry tombing, the emphasis should not be placed on total exclusion of the liquid through capping layers, but rather on allowing the landfill liquid to flow through the body as if the waste were a soil of low permeability. It must be remembered that the pollution levels of the leachate from treated waste should be significantly lower than that of untreated waste, although a degree of leachate treatment will still be required. The discharge of the leachate into a robust system, such as a constructed wetland, may be an indefinite solution to landfill closure. Finally, the treated leachate should be used for the wetting of the waste.

9.7 BIOGAS MANAGEMENT

The management of biogas that is released from treated waste must still be considered. Landfill gas collection and treatment through flaring is an established technology and is implemented at landfill sites in Durban. However, these processes can only be applied in cases where the methane production rate is sufficiently high. The use of biofilter and for the oxidation of methane through the activity of *methanotrophic* bacteria is a technique that may prove to be applicable for the treatment of the low volumes of biogas emitted from a landfill (Streese et. al., 2003). The application would be more efficient in shallower landfills, where the surface is proportionately larger allowing for a greater volume of biogas extraction. These methods should be investigated under the subtropical climate of Durban in order to establish their efficiency under prevailing climatic conditions.

If the biological oxidation is not sufficient for the treatment of gas, particularly the early emissions from waste that has undergone a shorter period of waste treatment, then active gas extraction may still be implemented.

9.8 DAILY COVER

The use of sand or similar material as a daily cover is not desirable for the proposed operations. This is due to the low permeability of the material and the fact that the material is inert implies that landfilling of inert sand and builder's rubble at a site designed to handle organic waste represents a waste of landfill airspace. Concessions have been made that allow for the use of organic material such as pine bark thus the use of composted material, such as the fine treated waste as a cover may be allowed.

9.9 COSTS

An extensive financial model of the operation including capex, operational and financing costs should be compiled if a full scale treatment operation is to be developed in Durban. However, for the purposes of this study and in order to gain an idea of the cost implications of the implementation of a full scale DAT project, a simplified approach has been adopted.

9.9.1 Cost of DAT system

Based on the figures given by Paar (2000), the cost of a DAT operation is estimated to be approximately €25/ton of waste treated.

9.9.2 CDM Finance

As discussed in chapter 2, the CDM is a possible means to help finance an MBT operation. The UNFCCC methodology **"Tool to determine methane emissions avoided from dumping waste at a solid waste disposal site"** is used to estimate the Certified Emission Reductions that would be earned if the Durban Waste Stream were to be treated.

The calculation (Appendix D) gives an answer of 0.43 CERs/ton of waste treated. Based on the current CER floor price of approximately €13.50/CER (www.pointcarbon.com), the potential CER revenues are €5.80/ton of treated waste.

9.9.3 Landfill Airspace Saving

The savings in landfill airspace is a significant consideration, with 50-60% savings reported in the literature. The cost of landfill airspace is approximately R22/m^3, depending on the depth of the site and the cost of the lining system. With currently landfilling methods achieving a placement density of approximately 1t/m^3, the unit cost of the airspace is approximately R22/t. If by treating the waste a 50% saving in landfill airspace is achieved, a net saving of R11/t is realized.

9.9.4 Total Cost of full scale DAT operation

Thus based on the current euro exchange rate of R13/€, the total cost of the DAT operation, including the financial benefit of landfill airspace savings and CER generation, is calculated as follows:

€25/t*R13/€ - €5.8/t*R13/€ – R11
= R238.6/t
≈ R240/t

Thus the cost of implementation of a full scale operation at the Bisasar Road Landfill Site is approximately **R110 000 000**.

By comparison, the cost of a high-tech plant would probably amount to approximately R1 billion.

An important consideration is that by saving landfill airspace, the landfills will remain operational for a longer period of time. This is of great significance when considering the difficulty in locating new sites. The NIMBY, "not in my back yard", attitude of the public results in pushing the location of the new site further from the urban areas and away from the populations which they serve. This has adverse effects on the cost of landfilling, in particular, the transportation of the waste to the site. This should be investigated as transportation costs already constitute approximately 70% of the metro's waste management budget, despite the central location of the Bisasar Road Landfill Site. It is therefore anticipated that the savings in terms of avoided transport costs could balance the cost of waste treatment.

9.10 SUMMARY OF OPERATIONS

The overall treatment and landfilling operation using the DAT system in conjunction with a MRF is summarised in Figure 9.1

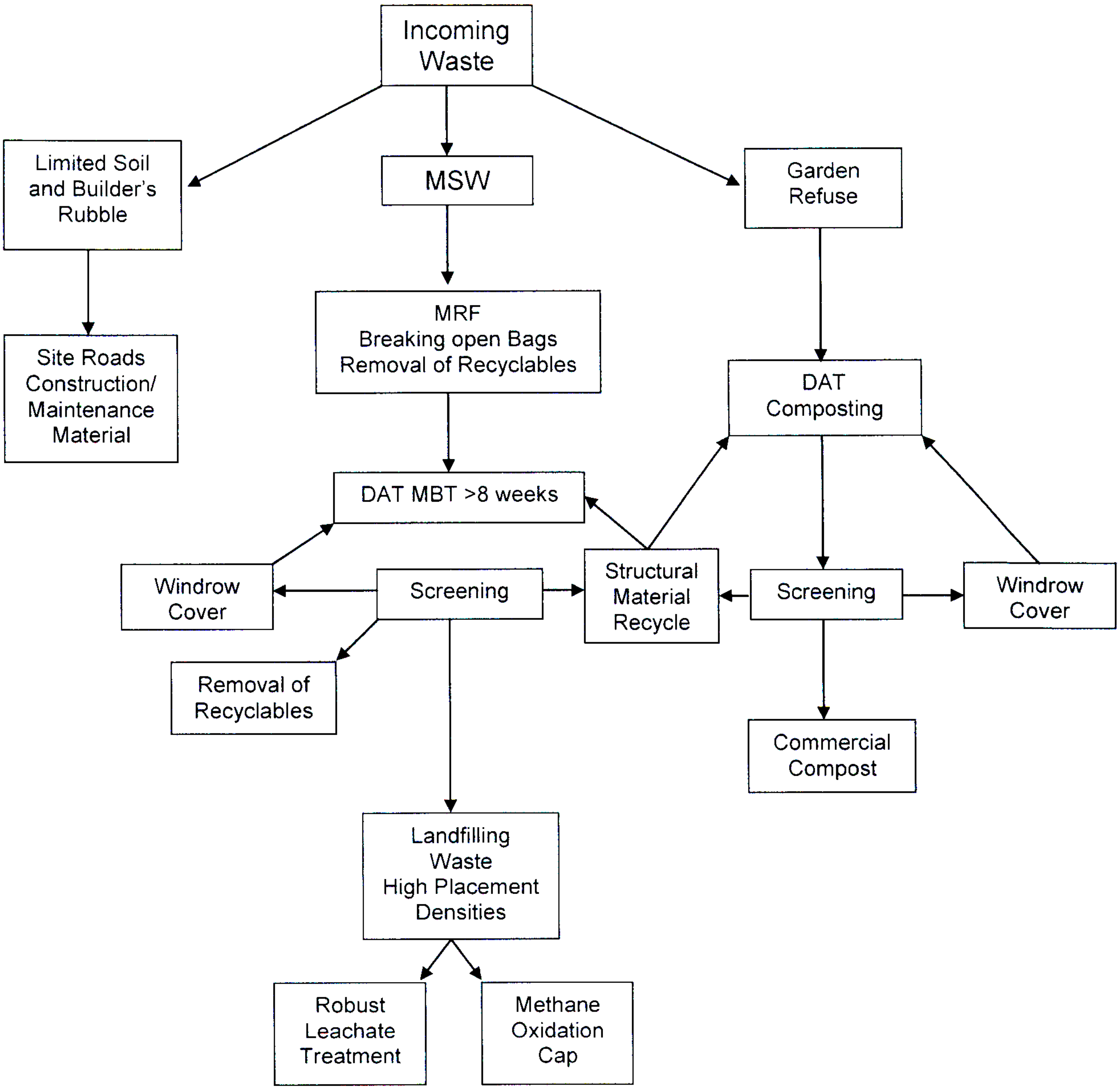

Figure 9.1 MBT and Landfill Operations Outline

9.11 COMMENTS ON RECOMMENDATION

The implementation of a waste treatment operation, as discussed above, will result in many social and environmental benefits. The leachate emissions will be significantly improved, however the contaminant concentrations will still be too high for discharge into the natural environment. Biogas generation shouldn't be an issue with bacterial methane oxidation used to treat the gas that is produced. The operation will add to the cost of waste disposal, although this is offset by the value in keeping the current sites operational. The landfill does not meet the criterion for sustainability as it is likely to require attention more than 40 years after closure. However, the reduction of the pollutant levels in the leachate, the saving in landfill airspace and the reuse and recovery of recyclable materials and compost all improve the sustainability of the landfilling operation.

CHAPTER 10

FINAL COMMENTS

Introduction

The outcome of this research has provided information on the mechanical biological treatment of municipal solid waste through the implementation of the DAT, the anaerobic behaviour and emission characteristics of this waste, the formulation of a emission model and has led to recommendations on the possible design and operation of a landfill that meets the criterion of sustainability. The findings of this research can also provide a technical framework for the development of a standard waste treatment operation in a context, such as South Africa or other developing countries, where the treatment of waste is not a regulation. This chapter serves to summarise these findings and assess the achievement of the objectives laid out in chapter 1, namely:

- To assess the viability of implementing the DAT under the sub-tropical climate of Durban and to offer recommendations for future local DAT operations;
- To investigate the emissions of treated waste once landfilled and to assess the benefit gained through waste treatment.
- To investigate the effect of screening the treated product on the emissions produced during anaerobic degradation of the treated waste.
- To model the pollutant release from the waste in order to assess the polluting potential of the treated waste, and the benefit of MBT over untreated waste.
- To use the findings of the study to recommend a waste disposal operation for the EThekwini Metro.
- The assessment of the Mechanical Biological Treatment of Waste as a Sustainable waste management solution

The Viability of Mechanical Biological Treatment using the DAT and Recommendations for future DAT operations

The use of standard landfilling equipment and implementation of the Dome Aeration Technology method of aerated windrows in Durban provided results comparable to those of the operations in Europe and particularly in Germany, at the Cottbus Landfill, where this technology was firstly applied. The use of a large proportion of slowly degradable material, such as the pine bark, indicated the requirement for an extended degradation process (longer than 4 months as in Cottbus) to allow for a greater degree of biological stabilization. Areas of poor ventilation, characterised by relatively high temperatures, low oxygen levels and detectable concentrations of methane, were also identified; suggesting variable efficiency in the biological stabilization within the windrow. Another important factor is the significance of windrow desiccation, and the importance of achieving the correct moisture balance during the preparation stage, although the irrigation of the window as a means to mitigate this should be studied further. The high oxygen concentrations measured from Windrow 3 indicates that the aeration supply surpasses the requirements for aerobic activity, and the conservative approach adopted in terms of window sizing and waste input ratios was not warranted. **Thus the construction of larger windrows, using less structural material is a viable option.**

The benefit of Waste Treatment on Landfill Emissions

Treatment of waste resulted in a significant reduction in peak COD and ammoniacal nitrogen loadings, with long term concentration loads of ammoniacal nitrogen far higher in the untreated waste. The treated waste did not exhibit an acidic inhibition of methanogenic and the COD loadings dropped to relatively low values within a short time period, as opposed to the untreated waste where an extended acidic stage produced highly polluted leachates. This has implications on the landfill aftercare, where a less intensive leachate treatment would be required for the liquid emissions from treated waste. Due to the poor performance of two of the DAT windrows the comparison of treatment time was not possible. However, it is reasonable to assume that with proper treatment, a longer treatment time should yield a more stable material than that of the 8 weeks treated waste. What is clear however, is the necessity to have a well controlled and monitored biological treatment process if the technology is to be applied locally. To summarise, the aerobic treatment of waste is of benefit in terms of landfill emissions. However, more studies are required to investigate the effect of the duration of the treatment on these emissions.

The Effect of Screening of Screening the Waste on the Anaerobic Emissions

The post treatment screening of the waste proved very effective in separating the high calorific fractions (paper and plastic) from the fine material. This has benefits if incineration or waste-to-energy schemes are to be adopted in South Africa without an established source separation scheme. In terms of the benefit to the DAT process, these fractions must be separated from the coarse waste stream if the structural material is to be recovered for reuse in the DAT operations. The small particle size of the majority of the fine fraction could negate the need for landfill cover and coupled with the high compaction rates reported by other authors this could result in significant savings of landfill airspace. There is no discernable difference between the coarse and fine material in terms of anaerobic emissions. In summary, **the screening of waste is**

beneficial in terms of removing low density, high calorific materials from the waste stream. Furthermore, the lab tests show that the **coarse fractions release more pollutants than the fine fractions. In the long term predictions from this research, however, there is little difference in the emissions of the two fractions once landfilled.**

The Modelling of the Pollutant Release

The development of leaching functions and emission models **was successful as a means to evaluate the difference in the waste tested during *this* study**, particularly the ammoniacal nitrogen model which is only affected by leaching and pollutant removal via leachate flux. The column test is well suited to the emissions model due to the dominance of the large leachate flux. In order to apply the model to full scale examples for COD modelling, a biological consumption component should be added. Further tests are required to improve the reliability and repeatability of the emission model if the results are to be used reliably at full scale.

Recommendations on Landfill Design and Operation

The findings of the literature review, DAT waste treatment study and anaerobic emission study **enabled recommendations to be made for the design and operation of a landfill site**. These recommendations are an aerobic pretreatment stage of 8 weeks, depending on space availability, and the landfilling of waste to high compaction densities with a biological methane oxidation capping system at an additional cost of approximately R240/t. This excludes the savings in transportation to new sites, which should form part of a thorough financial assessment.

Conclusion

In conclusion the achievement of a sustainable landfill in which emissions were of acceptably low values after forty years of aftercare does not appear to be viable, however this does not mean that the benefit is of no use. It is true that the ultimate focus should be on following the hierarchy of waste management, and waste reduction, reuse and recycling should be the priority. However, there will still be the need to dispose of the residues of these processes or of material that cannot be used or processed and the landfill will be required to serve this need in the foreseeable future.

The ultimate finding of this research is that although the Mechanical Biological Treatment does not present the ultimate solution to achieving the objective of a sustainable landfill, it does constitute **a positive step in the move towards sustainability**.

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APPENDIX E

JOURNAL PUBLICATIONS

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Country Report

Introducing mechanical biological waste treatment in South Africa: A comparative study

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Abstract

This paper presents the results of the first pilot project on mechanical biological waste treatment (MBWT) in South Africa. The study has shown that biological waste treatment in windrows using a passive aeration system that utilises thermal convection to drive the aeration process within a windrow of waste is appropriate for South Africa, in relation to low capital costs, low energy inputs, limited plant requirements and potential for labour-intensive operations. The influence of climate, waste composition and operational facilities was evaluated to optimise the treatment technique to local conditions. The maximum temperatures reached during the intensive thermophilic stage were effectively equivalent to the German experience. The lower CO_2 production experienced in the South African trials was attributed to a different waste stream (high presence of plastics) due to the absence of a proper source separated waste collection system. An accurate adjustment of the input material (structural matter in particular) to the specific ambient conditions and irrigation during composting should result in higher organic carbon degradation efficiency in equivalent timeframes. This preliminary experience suggests that the applicability of MBWT in emerging countries, such as South Africa, is directly dependant on the mechanical treatment steps, available operational facilities and nature of the input material.

1. Introduction

Current South African legislation allows for landfilling of untreated waste and focuses on the "concentrate and contain" approach, which results in a long-term environmental risk. The Polokwane Declaration, signed in 2001 during the First South African Waste Summit, sets new standards towards "the reduction of waste generation and disposal by 50% and 25%, respectively, by 2012 and the development of a zero waste plan by 2022". The stress towards a "sustainable landfill" is now being refocused to the implementation of an integrated waste hierarchy based on increasing desirability of the outcome and decreasing waste volumes: waste reduction being the most desirable action, followed by reuse, recycling, composting, treatment and disposal. The introduction of mechanical biological waste treatment (MBWT) prior to disposal would be beneficial to waste recycling whereby the mechanical process allows for material recovery, while the aerobic stabilization of organic waste has potential for landfill emission reduction. A comparative analysis of available waste treatment options indicated that not all of the existing systems are appropriate for South Africa, as high capital and running costs make some technologies unviable, particularly for application in rural areas (Griffith, 2005). Factors such as low-cost infrastructure, low energy input, the potential for labour-intensive operations, job creation and poverty alleviation, and application in daily landfill operations were considered minimum requirements.

MBWT in self-aerated windrows using the dome aeration technology (DAT) was selected as a suitable option for the first pilot project in South Africa using full-scale

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windrows, initiated by the University of KwaZulu-Natal in collaboration with the eThekwini Municipality (waste disposal unit) and the University of Dresden (Germany). Durban was selected for this pilot project because it is representative of waste management in South African metropolitan areas, with respect to nature and volume of waste, refuse disposal facilities and minimisation strategies. The disposal of waste in South Africa is regulated by the minimum requirements for disposal of waste by landfill (DWAF, 1998) where design and operation of waste disposal facilities is based on the nature of the waste (general or hazardous), the size of the waste stream and on the potential to generate significant volumes of leachate in relation to the local climatic water balance. Durban, located on the subtropical eastern coast of South Africa, is the largest city in the province of KwaZulu-Natal. The eThekwini Municipality, which includes Durban and the surrounding areas, has a population of over 3 million people (2001 Census) and generates approximately 1.2 million tonnes of municipal solid waste (MSW) per annum. The main economic activities are related to manufacturing, tourism, finance and transport. The waste stream analysis for the Durban area reports the following material proportions: hard plastics: 6.4%; soft plastics: 11%; glass: 7.1%; tin/aluminium: 6.9%; cardboard: 9%; paper: 10.3%; organics: 42.5%; and other: 6.8% (SKC Engineers, 2002). Although garden refuse is collected separately, it is also disposed of at MSW landfill sites. The eThekwini municipality operates three active general waste sanitary landfills through a service unit called Durban solid waste (DSW). The oldest and largest active site, the Bisasar road landfill site, is a 44 ha general solid waste site located 7 km from central Durban; it was established in 1980 and has been managed by DSW since 1995. The landfill site receives mainly domestic, commercial and construction refuse, averaging around 3000 tonnes per day; equating to 650,000–750,000 m^3 per annum.

The main objective of the South African pilot project was to optimize the size of the windrows in relation to local factors such as the climate, waste composition and available operational facilities. This paper reports on a comparative analysis of the performance of the Durban windrows with the German experience.

2. MBWT pilot project rationale

The dome aeration technology (DAT) (Mollekopf et al., 2002) is a passive aeration system that utilises thermal convection to drive the aeration process within a windrow of waste. DAT is a unique self-aeration windrow technology and should not be confused with "The Chimney Process" proposed by Spillman and Collins (1981). The principle of the DAT method is the creation of large voids in a windrow of waste using metal structures, named domes and channels. Domes are positioned centrally in the windrow to allow for venting of the hot gasses generated by the degradation reactions through the chimneys and channels placed across the base of the windrow for drawing in ambient air. A large number of smaller voids in the waste are created by mixing in structural material (SM), which consists of rigid waste such as timber planks and branches.

Stage 1 of the pilot project involved the construction of three full-scale DAT windrows at the Bisasar road landfill in Durban in order to study the efficiency of the process for different composting times (8 and 20 weeks). Construction and operation techniques of the windrows adopted in Durban were based on the full-scale windrow sizes and geometry adopted at the Cottbus landfill in Germany, where the DAT was used for MBWT with a throughput of more than 50,000 tonnes per annum until 2005. Since 2005, DAT can be applied to post-treatment of residues from high-tech MBWT (with refractory-organic fine fraction remaining) to achieve waste stabilization in accordance with EU standards in only 3–4 months, with no turning (Mollekopf et al., 2002; Paar et al., 1999a,b).

The main factors that were likely to affect the performance of the Durban MBWT DAT windrows were predicted to be the subtropical climate, the waste stream and equipment availability. The construction rationale was based on the experience gained during test trials conducted at the Plauen landfill and later on full scale at the Cottbus landfill, in Germany, where MBWT has been conducted since 2000 in DAT windrows with the following dimensions: up to 110 m length, 11 m width and 3 m height (including 0.5 m of cover material), containing 22 domes and 46 channels (Paar et al., 1999a,b). The Cottbus site operations entailed a three step mechanical pre-treatment stage of mixing, shredding and wetting, followed by composting for 4 mo (Paar et al., 1999a,b; Mollekopf et al., 2002). The windrows can be constructed using standard landfilling equipment and their application into daily landfill operations is well detailed in Mollekopf et al. (2002), Griffith (2005), Trois and Polster (2007). During the mechanical treatment, general MSW and bulky waste (structural material – SM) is loaded into a shredder simultaneously in a volumetric ratio of 1:1 (2:1 by mass). The moisture content was adjusted to approximately 55% for optimal microbial activity. The adequate provision of moisture is considered crucial to the process as the DAT windrows are left undisturbed throughout the composting period.

2.1. Effect of the climate

Durban's climate is typically subtropical: warm and humid with a mean annual temperature of 20.5 °C and average annual precipitation of approximately 1000 mm (concentrated between October and April). Daily minimum temperature averages are between 21 °C in summer and 10 °C in winter. Maximum temperatures average around 30 and 17 °C in summer and winter, respectively. Relative humidity is very high and often exceeds 70% in the summer. The warm climate results in accelerated biological activity in landfills with significant biogas production during a period of 3–6 months after placement of the waste (Trois et al., 2001). Since the DAT flow mechanisms rely on the thermal gradient between the ambient air and the gas generated within the windrow, it was assumed that higher ambient temperatures could adversely affect the aeration rates. To optimise the size of the Durban DAT windrows, the ratio between the exhaust gas velocity experienced in Durban and in Cottbus/Plauen was modelled in relation to a spectrum of typical ambient subtropical temperatures (Fig. 1). Eq. (1), based on Bernoulli's energy equation, was used to estimate the effect that higher ambient temperatures would have on exhaust gas flow velocity (v) from the windrow, where $\Delta\rho/\rho_d$ = gas density gradient, h = height of the chimney, K = coefficient of energy loss, and g = acceleration due to gravity. For simplicity the contribution of the wind was excluded.

$$v = \sqrt{\frac{2g\left(\frac{\Delta\rho h}{\rho_d}\right)}{1+K}} \tag{1}$$

If the unknown parameter for energy losses (K) is assumed to be constant for all windrows, the flow velocity can be determined for different conditions as in Eq. (2), where T_1 and T_2 are different ambient temperatures, M_a and M_d are the molecular weight of the ambient air and dome gas, respectively, T_d is dome gas temperature and ρ_{d1} and ρ_{d2} are dome gas densities at T_1 and T_2, respectively.

$$\frac{v_{T_1}}{v_{T_2}} = \sqrt{\frac{\left(\left(\frac{M_a}{T_1} - \frac{M_d}{T_d}\right)h/\rho_{d1}\right)}{\left(\left(\frac{M_a}{T_2} - \frac{M_d}{T_d}\right)h/\rho_{d2}\right)}} \tag{2}$$

An ambient temperature $T_2 = 0$ °C was used as the reference value, due to the low temperatures experienced in Cottbus/Plauen. The ambient temperatures compared to this value were 10, 20 and 30 °C so to cover the expected seasonal range for the Durban area. The percentages of flow velocity at given temperatures (V_{T_1}) to the flow velocity at the reference temperature are plotted against windrow temperature in Fig. 1. Dome oxygen and carbon dioxide concentrations of 15% and 5%, respectively, were used to calculate M_d. Fig. 1 shows that the highest temperature ranges, expected during the intense biological activity, which requires the highest flow rate, results in the smallest decrease in relative flow velocity. In Cottbus/Plauen, the upper range value of required oxygen concentration (15%) is reached after approximately 15 days at a temperature of 60 °C (Paar et al., 1999a,b). At 60 °C, the flow velocity expected in Durban would be between 10% and 30% less than that experienced in Cottbus/Plauen on the assumption that ambient temperatures in Durban would be 10–30 °C higher.

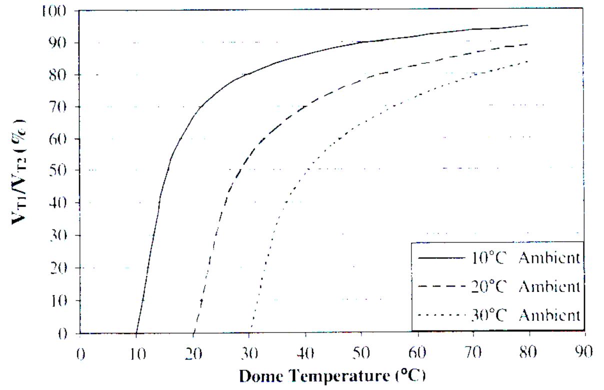

Fig. 1. Percentage of flow velocity at given ambient temperatures to flow velocity at 0 °C calculated from Eq. (2).

Assuming that the windrow temperatures remain above 40 °C for the entire composting period, the chimney flow velocity expected in Durban would be between 15% and 50% less than Cottbus/Plauen. The oxygen content measured in the Cottbus/Plauen windrows remained well above the recommended minimum oxygen levels of 10–15% (EPA, 1995), suggesting that there is a certain degree of flexibility in terms of lower aeration rates. It was anticipated that during the Durban summer, when ambient daytime temperatures can reach 30–40 °C, the oxygen contained in the voids within the windrow would be sufficient to supply the lower rate biological activity that is expected in mesophilic conditions (<45 °C), therefore minimising the negative effect of very low airflow until the ambient temperatures dropped, restoring the thermal gradient. On the basis of the anticipated average ambient temperature difference of approximately 20 °C, a conservative approach was adopted in the construction of the Durban windrows which were 20% smaller than in Cottbus/Plauen.

2.2. Influence of the waste stream

Since no source separated collection is performed in Durban, the supply of suitable structural material (SM) proved to be a limiting factor in windrow construction. Operations could only proceed once a suitably large stockpile had been accumulated. This is unlike the Cottbus procedure, where bulky waste is separated at source and then homogenised during the mechanical treatment. In Durban, dry branches (10–200 mm in diameter) were selected while green and leafy material was rejected as this would contribute to the more readily degradable fraction of the input waste. Chipped pine bark (PB) was used to cover the windrows to insulate against heat and moisture loss and to minimise odour. Although the MSW to SM ratio (by mass) used in Cottbus/Plauen was 2:1, a conservative approach of 1:1 was initially used in Durban to provide more voids and a lower oxygen demand and thus ensure a completely aerobic waste body. After the construction of windrow 1, PB was mixed with the waste to supplement the SM and to serve as moisture sink over the composting period. The final waste input ratio for windrows 2 and 3 was 2:1:1 – MSW:SM:PB.

2.3. Equipment and construction technique comparisons

The pre-treatment steps in Cottbus/Plauen, and the alternatives employed in Durban are summarised in Table 1. For windrow 1, the waste was initially crushed by repeated passes of a landfill compactor. For windrows 2 and 3, lightly shredded MSW was used from Faun Roto-Press collection trucks. An excavator was used to break up the large fractions of structural material. Waste was laid on a large area in a thin layer and wetted by repeated passes of a water truck to increase the initial moisture content to 50–60%. Further crushing of the waste was thus also achieved during the wetting step. For windrow 1, a front-end-loader created one stockpile of material from separate stockpiles of MSW and SM and then turned it, mixing the two input materials. For windrows 2 and 3, an excavator was used to load the three different material streams (MSW, PB and SM) into an articulated dump truck (ADT) so that it was sufficiently mixed after tipping. The windrows were constructed on the surface of the landfill site, which consisted of a soil capping. A summary of the construction parameters adopted in Durban and in Cottbus is presented in Table 2. Fig. 2 shows the typical layout of DAT windrow 3 constructed in Durban. The cost of the manufacture of the domes and channels equates to approximately US $3.5/m^3$. Although the DAT structures can be reused, a 30% loss was experienced due to damage while dismantling, although this could be reduced with improved operator experience. A summary of the cost of the plant used in the DAT operation is shown in Table 3.

Table 1
Comparison between mechanical equipment used in the Cottbus/Plauen and Durban trials

Pre-treatment step	Cottbus/Plauen equipment	Bisasar road alternative
Mixing	During shredding stage	Excavator Pay-loader
Shredding/ crushing	Slow revolving shredder	Landfill compactor Excavator Passage of water truck Roto-press refuse trucks
Wetting	Bar sprinkler	Water truck (20,000 ℓ)

3. Process performance monitoring

System performance was quantified by analysing the degraded organic matter (material balance) and the development of the rotting process. For windrow 1 only the evolution of moisture content was measured by means of in-situ Time Domain Reflectometer (TDR) Probes commonly used to measure soil moisture content; however, these results are not presented herein. For windrows 2 and 3, the exhaust-gas-composition, temperature and velocity of the emitted gasses were measured in the chimneys; gas-composition and temperatures at selected locations on the windrow were also recorded; every day for the first week and once a week thereafter for 20 weeks. The nature of the flow paths through the DAT windrow results in areas of poor and areas of good ventilation and probes were positioned in order to assess the variations within the windrow (Fig. 2). Conditions in the waste body were analysed using permanent perforated 2 m PVC probes, inserted in the well and poorly ventilated areas as shown in Fig. 2. The windrow body and the exhaust gas temperatures were measured by inserting a thermocouple (Major Tech MT-630) into the probes (at 0.5, 1 and 2 m depths) and into a hole in the chimney. Ambient temperatures were obtained from a weather station located at the Bisasar Road and from the South African Weather Bureau for the Durban area. Concentrations of O_2, CO_2 and CH_4 in the exhaust gas (% in air) from the chimneys and probes were measured once a week, using an infrared gas analyser (Type GA94 – Geotechnical Instruments). Exhaust gas velocity and wind velocity were also measured using an anemometer (Silva Windwatch). The efficiency of the stabilisation process was assessed by characterising the treated waste after 8 and 16 weeks of composting, together with the input material. The characterisation was performed using standard eluate tests in distilled water with a liquid to solid ratio of 1:10. After 24 h, the eluate produced was filtered with a Whatmann 40 filter paper and analysed for the following parameters: TS, VS, TOC, TKN, NH3, NOx, pH, COD, BOD, (ASTM, 2004). TS and VS, as well as moisture content and respiration index (4 days) tests, were performed on the solid matter according to standard methods (ASTM, 2004) and EU directives (Anonymous, 1999).

Table 2
Summary of the windrow construction parameters as adopted in Durban and in Cottbus

	Windrow 1	Windrow 2	Windrow 3	Cottbus test windrow
Waste input (ratio)	MSW:SM (1:1)	MSW:SM:PB (2:1:1)	MSW:SM:PB (2:1:1)	MSW:SM (2:1)
No. of domes	3 Domes waste 3 Domes bark	3 Domes	4 Domes	8 Domes
Volume	325 m^3 MSW – SM 325 m^3 Bark	350 m^3	450 m^3	950 m^3
Time required for construction	20 h	20 h	12 h	–

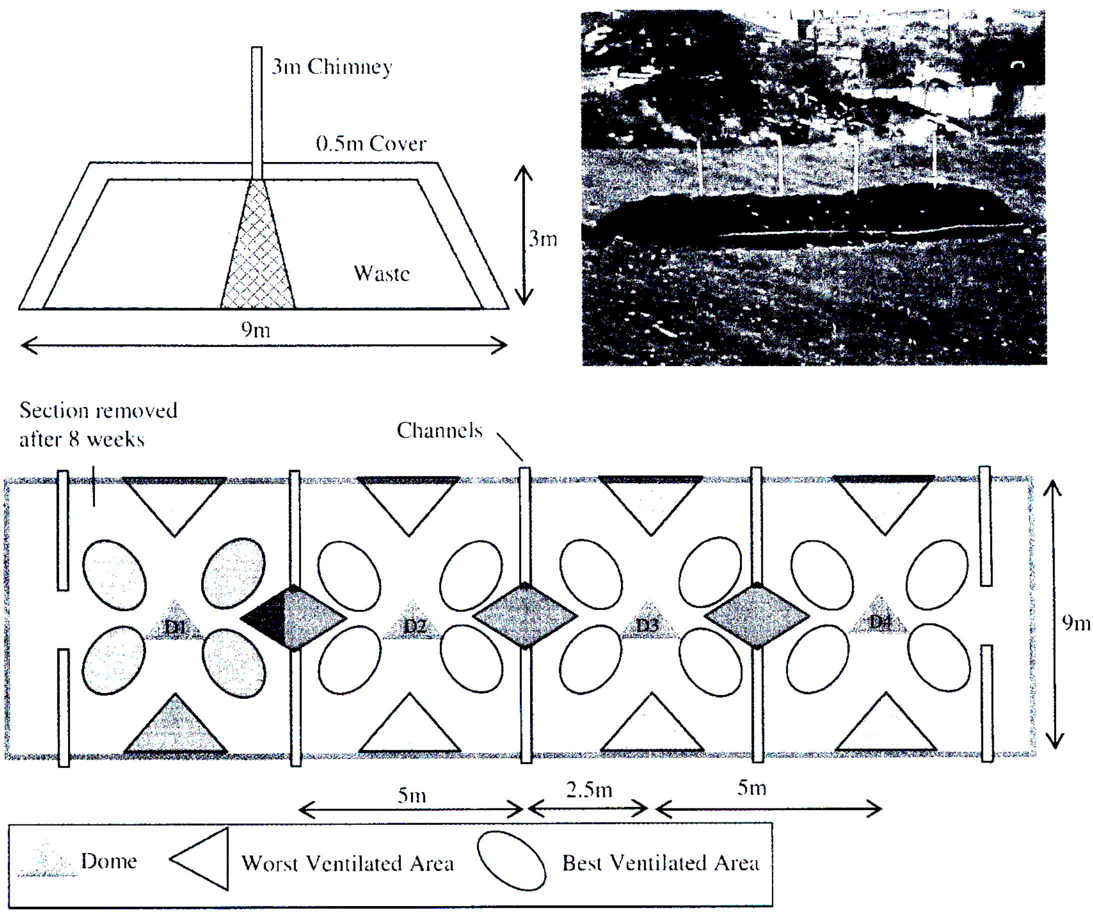

Fig. 2. DAT windrow cross section and plan showing poor and well ventilated areas. The plate shows a completed windrow at the Bisasar road landfill site in Durban.

Table 3
Summary of the costs of the Durban trials

Machinery/ labour	Operation cost US$/h	Effective hours per day	Amount of wastes (t/day)	Operation cost US$/ ton
Truck (ADT)	35	4	60	2.35
Front-end loader	45	1		0.75
20KL water tanker	30	2		1.00
Excavator	45	6		4.50
General labourers	2.5	8		0.35
Total cost				8.95

4. Results

4.1. Waste characterisation

Table 4 shows the results of the characterisation of the input and output material (after 8 and 16 weeks of treatment) during the Durban trials in comparison with the German operations and standards. Accuracy checks were conducted throughout the experimental phase; the average values and standard deviation reported in Table 4 were calculated from the results of 4–6 eluate batches on each sample. The results show that the treatment does reduce waste pollutant loading in terms of COD, BOD and NH_3. However, it must be noted that the COD values for the 16 weeks treated waste was higher than that of the 8 weeks treated waste. This would suggest inefficiency in the process, and it is likely due to desiccation considering the low moisture content of the 16 weeks waste (approximately 17%). Eluate tests of material from windrow 3 at the end of the composting process were not conducted.

4.2. Process performance

The performance of the Durban windrows 2 and 3 are compared with an average of the results of the Plauen trials (Paar et al., 1999b) in Figs. 3–6. The evolution of dome temperatures and oxygen content with time for windrows 2 and 3 and the Plauen trials is shown in Figs. 3 and 4, respectively. From Fig. 3 it can be observed that all three

Table 4
Characterisation of the windrow material before and after composting during the Durban and the Cottbus trials, in relation to the German standards for MBT waste

Eluate test	German standards	Durban trials – MSW			Cottbus/Plauen trials – MSW	
		Untreated	8 weeks windrow 3	20 weeks windrow 2	Untreated	24 weeks
pH	5.5–13.0	5.29 ± 0.15	6.83 ± 0.1	7.14 ± 0.14	–	
COD (mg/l)	–	8508 ± 1617	2437 ± 40	2642 ± 231	–	–
BOD (mg/l)	–	2711 ± 637	529 ± 36	469 ± 57	–	–
NH_3-N (mg/l)	<200	84.3 ± 33	17.8 ± 3.6	1.14 ± 0.5	–	–
NO_x-N (mg/l)	–	71.6 ± 9	5.7 ± 1.1	<0.05	–	–
TKN (mg/l)		472.81 ± 24	–	48.68 ± 3	–	–
TOC (mg/l)	<250	1560 ± 78	790 ± 40	646 ± 32	3600[a]	200[a]
Solid matter						
TS (%)	–	42.3 ± 14	64 ± 3	82.2 ± 4	45[a]	70[a]
VS (% TS)	–	61.1 ± 7	25.6 ± 1	32.5 ± 2	–	–
RI_4 (mgO_2/gTS)	5	16.7±	–	9.3±	60[a]	6[a]

[a] Note that during the Cottbus/Plauen trials only the parameters required by the EU regulations for MBWT were determined.

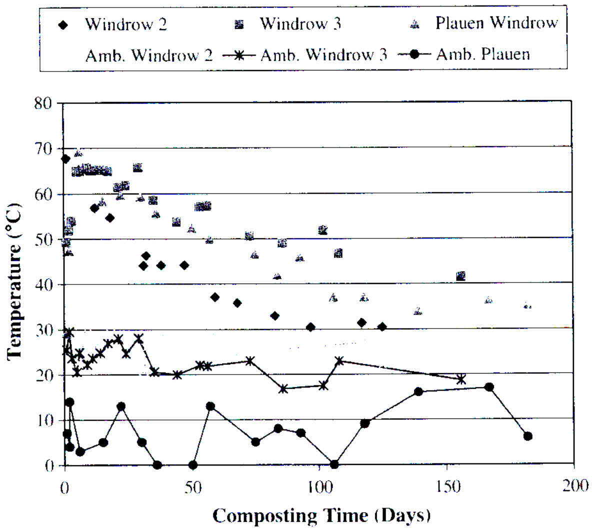

Fig. 3. Comparison between the temperature variations in the exhaust gas in windrows 2 and 3 in Durban and an average of the Plauen trials. Ambient temperatures are shown.

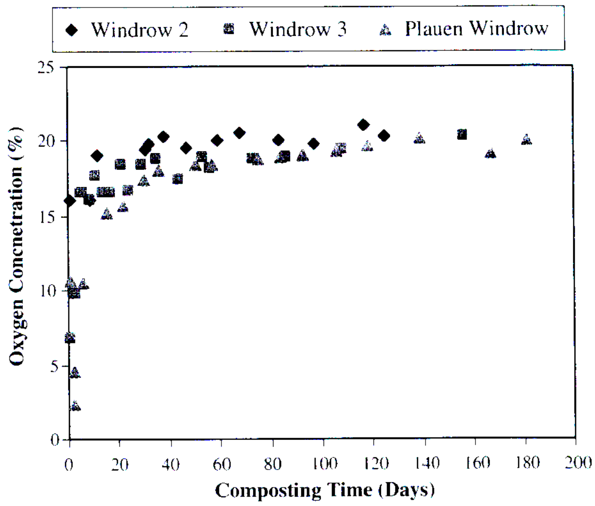

Fig. 4. Comparison of O_2 concentration in the exhaust gasses from windrows 2 and 3 in Durban and an average of the Plauen trials.

windrows reach approximately the same peak temperatures in similar timeframes.

The temperature of windrow 2 decreases more rapidly than the other two while the temperatures of windrow 3 remained comparable to the Plauen windrow. The peak temperature difference between the ambient air and the windrow dome of approximately 45 °C for the Durban windrows is 20 °C lower than that experienced during the Plauen trials. Although this would suggest a lower rate of aeration for the Durban windrows, analysis of O_2 levels in the dome exhaust gas, as presented in Fig. 4, shows that sufficient oxygen was present throughout the composting period. Fig. 4 shows that windrow 2 reached higher oxygen concentrations in the shortest period of time. The oxygen concentrations for both Durban windrows were equivalent to Plauen for the majority of the composting period. Oxygen concentrations remained close to atmospheric levels in the exhaust gas of windrow 2, so as to suggest a lower rate of biological activity due to early desiccation. The low temperatures and high oxygen concentrations in windrow 2 are indicative of low biological activity at a relatively early stage. This could be attributed to low putrescible content, but is more likely to be as a result of dessication, as indicated by the low moisture content. The higher temperatures and lower oxygen levels in windrow 3 show that, unlike windrow 2, windrow 3 remained biologically active for the duration of the test suggesting a better overall performance. Fig. 5 shows the evolution of the chimney flow velocities in windrows 2 and 3 in comparison with the German trials.

Fig. 5 shows that windrow 2 experiences the highest chimney flow velocities, which correspond to the highest airflow. The flow velocity of windrow 3 is close to that of the Plauen trials, although the results from windrow 3 show more fluctuation due to the effect of wind. The reason

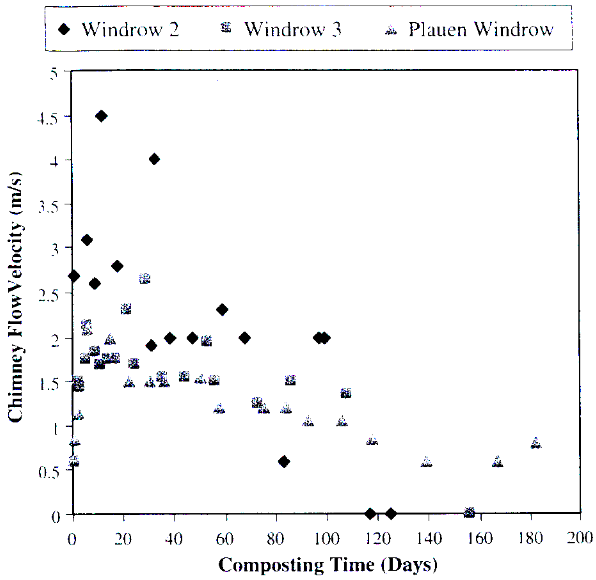

Fig. 5. Comparison of chimney flow velocities from windrows 2 and 3 in Durban and an average of the Plauen trials.

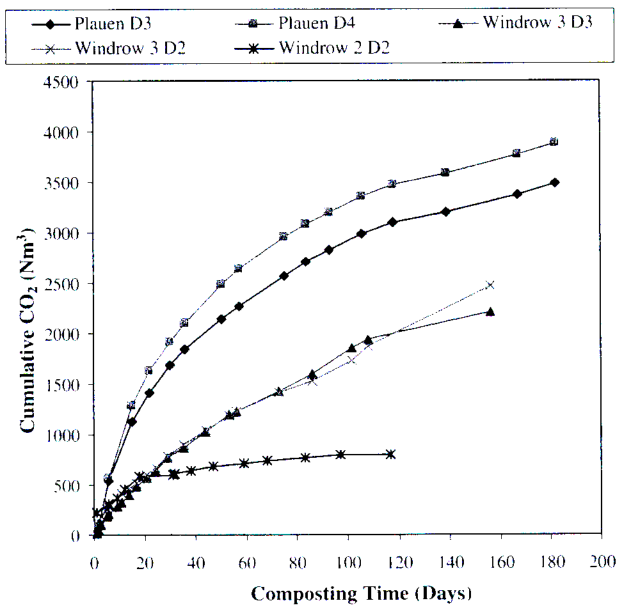

Fig. 6. Comparison of CO_2 production from windrows 2 and 3 in Durban and an average of the Plauen trials.

for windrow 2 experiencing the highest flow rate despite the lowest temperature difference is most likely due to the suction effect of the wind. The composting period for windrow 2 falls within the season that experiences higher wind speeds than that of the compost period for windrow 3. Furthermore, the waste in windrow 2 may have been more porous, allowing for easier airflow. The higher airflow in windrow 2 would have been a contributing factor in the rapid moisture loss.

4.3. *Estimated organic conversion*

A means to evaluate the degradation performance of the windrows is by calculating the total CO_2 produced during the composting period. The biodegradable organic carbon orgCb was estimated theoretically for the windrow input material used in Durban according to Table 5. Based on the waste stream analysis (SKC Engineers, 2002), 50% of the orgCb is putrescible while the remaining fraction is slowly degradable (wood, pine bark, mixed paper).

The measured CO_2 production gives an indication of the aerobic catabolism of putrescible organic materials as described by the general stoichiometric Eq. (3).

$$C_aH_bO_cN_d + eO_2 \rightarrow xCO_2 + yH_2O + zNH_3 \quad (3)$$

The theoretical maximum CO_2 production based on the dimensions of the Durban windrows is 4930 Nm^3. The measured cumulative CO_2 production in Durban and Plauen is presented in Fig. 6. After 20 weeks of composting in windrow 2, the cumulative CO_2 released was 480 Nm^3, indicating a low catabolism of putrescible matter (less than 10% of the biodegradable organic carbon). For windrow 3, 8 weeks of composting resulted in the release of approximately 1200 Nm^3 of CO_2. After 20 weeks, 2300 Nm^3 CO_2 equivalent to 47% of the theoretical biodegradable content was converted into CO_2 through catabolic reactions. The conversion of a significant part of the slowly degradable fractions into stable humic fractions also occurs (Cappai et al., 2005), but was not measured during this study.

5. Discussion

Overall, windrow 3 was the most efficient of the Durban windrows in comparison with the German trials and in

Table 5
Biodegradable organic carbon per m^3 of windrow input material (Trois et al., 2005)

Input material	Mass per m^3 kg	Moisture content w%[a]	Dry DM	Mass orgC/kg DM	f_{bi}[a]	orgCb per m^3
Unit	kg	%	kg	kg		kg
Putrescibles (food waste)	106.25	60	42.5	0.42	0.8	14.12
Mixed paper	48.25	8	44.4	0.25	0.5	5.65
Wood	83.3	20	66.7	0.23	0.5	7.60
Pine bark	166.7	47	88.3	0.13	0.25[b]	2.81
Total						30.19

[a] Proportion of organic material that is degradable. Source: Cossu et al. (1996).
[b] Estimate based on 60 day soil incubation test in Haug (1993).

relation to process performance, carbon reduction and stabilisation kinetics. The higher ultimate CO_2 production from the Plauen DAT trials possibly reflects the difference in material input between Germany and South Africa, and the higher ratio of MSW used in the German windrow. Additionally, the German windrows were 20% larger, containing more material per dome. The absence of a proper at-source separation of the slowly or non biodegradable matter in the Durban waste stream is a potential cause for the slower degradation rates and lower overall CO_2 production. Other causes may be inefficient shredding during the Durban trials resulting in a smaller surface area for biological activity. A larger amount of readily degradable organics in the German trials is evident in the rapid onset of the CO_2 production in the earlier stages. Although CO_2 production and process kinetics were comparable, it is evident that the degradation processes in windrow 3 were still relatively active at the end of the trials (200 days) by comparison to the German counterpart. This is substantiated by the higher temperatures recorded for windrow 3 and is probably due to the continued breakdown of the large proportion of slowly degradable organics although the higher ambient temperatures may also be responsible. An optimum average concentration of oxygen ranging around 15% after the first week of composting (minimum range 10–15% from EPA (1995) and Williams (1998)) was measured throughout the rotting period, in each windrow. Therefore, ignoring localised areas of poor ventilation, the aeration rate surpassed the process requirements for the windrows used, suggesting that a larger windrow with a higher MSW to SM ratio can be adopted without affecting the efficiency.

Windrow 2 experienced an early desiccation and high aeration rates. This suggests that despite a conservative approach being adopted during the wetting stage, the losses associated with wetting using a water truck and not irrigating the windrows during composting can significantly affect the overall performance. Windrow 3 remained biologically active for the entire composting period, although some areas experienced sustained high temperatures, coupled with low oxygen concentrations. These poorly ventilated areas can be attributed to inadequate mixing of the structural material, localised compaction during the windrow construction and settlements after construction of the window.

6. Conclusions

Based on the results from windrow 3, the DAT proved to be a successful method of providing oxygen to an aerobic biodegradation process. The maximum temperatures reached during the intensive thermophilic stage were effectively equivalent to the German experience, and a reduction in aeration due to higher ambient temperatures was not apparent. The CO_2 production was comparable between windrow 3 and the German windrows when the differences in waste composition and windrow size are taken into account. The conversion of 47% of the degradable carbon into CO_2 indicates a high reduction of the putrescible content. The transformation of slowly degradable organic material into stable humic forms must be remembered and thus the overall transformation of organic carbon may be substantially higher than the 47% indicates, as suggested by other authors. The slowly and non biodegradable material present in the Durban waste stream (e.g., plastics, glass and metals) should be separated from the waste stream before treatment in order to improve the process efficiency. A poor mixing and shredding phase, due to unsuitable facilities, seems responsible for the occurrence of poorly ventilated areas, dry niches and preferential aeration pathways. The use of standard landfilling equipment proved to be appropriate in the mechanical preparation and subsequent construction of the DAT windrows at pilot scale, but the energy efforts may be too high for full scale operations. Although the raw materials for the DAT structures should be widely available, the application of the DAT to sites where equipment is limited will be difficult and may be a problem in other developing countries where waste management infrastructure is not well established as in South Africa. The advantages of using an available landfill facility are significant for smaller landfill sites, in rural or peri-urban environments; however, the unavailability of suitable structural material may be detrimental to full-scale implementation. One potential solution is to use dry garden refuse as structural material while screening the composted product, as implemented in Cottbus, in order to recover such material as soil conditioner or cover material for other windrows. Additionally, in order to avoid desiccation (as experienced in windrow 2), a mitigation measure may be to control the windrow airflow after the early oxygen intensive thermophilic stage by restricting the chimney area or irrigating the pile as proposed by Münnich et al. (2006).

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Long-term emissions from mechanically biologically treated waste: Influence on leachate quality

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Abstract

Long-term emissions from municipal solid waste landfills can be reduced by mechanical-biological treatment (MBT) of waste prior to disposal. The pretreatment accelerates waste degradation resulting in a reduction of the landfill's polluting potential. This study reports on the applicability and efficiency of MBT in Durban, South Africa. Waste treatment in passively aerated open windrows, using the Dome Aeration Technology (DAT), was identified as an appropriate technology due to low construction and operational resource requirements. Three self-aerated windrows were set up at the Bisasar Road Landfill in order to study the efficiency of the process for different composting timeframes (8 and 20 weeks). The 'post-landfilled' behaviour of the pretreated material was analysed in anaerobic lysimeters. The effect of different degrees of degradation was studied in relation to waste composition and rate of irrigation. The lysimeter tests demonstrated that the initial acidic inhibition that is characteristic of waste with high organic content can be eliminated through pretreatment. Notwithstanding the rapid onset of methanogenesis, high COD concentrations of non-degradable organics in the leachate remain after 200 d of testing. Despite the high COD levels, a clear benefit of waste pretreatment is the low concentration of ammoniacal nitrogen after only 8 weeks of composting. The results of this research can be used to define a framework for sustainable waste disposal, particularly in relation to the subtropical climatic conditions experienced in Durban, resource availability and waste composition.

Keywords: mechanical-biological treatment (MBT), landfills, MSW, leachate

Introduction

As a developing country, South Africa faces the challenge of meeting international standards in service delivery. Waste disposal, in particular, must be environmentally acceptable and economically sustainable. Currently, 95% of waste is disposed in landfills (DWAF, 1998a). It is well accepted that landfilling of large quantities of degradable organic material in a complex and heterogenic anaerobic environment, such as a landfill, will result in the formation of an inefficient biological reactor with the potential to produce persistent liquid and gaseous emissions. The significance of this problem has been recognised by the Department of Water Affairs and Forestry (DWAF) and the introduction of the *Minimum Requirements for Waste Disposal by Landfill* in 1994 was a crucial step in implementing more stringent landfill engineering guidelines to curtail long-term environmental impacts. The aim of the Minimum Requirements is to control landfill emissions through the 'concentrate and contain' approach, where the landfill is designed and operated as a multi-barrier system (DWAF, 1998a) in areas where significant leachate generation is expected, such as Durban. It must be remembered that in some of the drier regions of the country, leachate generation is sporadic or negligible (DWAF, 1998a).

A significant aspect of landfill management is the 'after-care' period when, after closure, control of landfill emissions is still required. The reduction of pollution levels from landfills is dependent on the stabilisation of the degradable organic fraction in the waste and the removal of the soluble pollutants through a combination of biochemical reactions and physical leaching. These processes are directly related to the flux of moisture through the waste body; the very same process that current South African legislation aims at reducing. The multi-barrier landfill, may, therefore, constitute an efficient short-term solution but can prolong the long-term pollution risk indefinitely. This fundamental truth on the nature of modern landfills is well accepted by the scientific community (Robinson, 2000; Cossu et al., 2003). In Europe the focus has now shifted towards the stabilisation of waste prior to landfilling via thermal (incineration) or mechanical biological treatment (Stegmann, 2005). *The European Council Landfill Directives 1999/31/EEC* (LFD) require member states to only landfill wastes that have been subjected to prior treatment (Robinson et al., 2005a).

A comparative study of available waste pretreatment techniques was initiated in 2002 (Griffith, 2005) to identify suitable solutions for South Africa that could be implemented at national level into established waste management systems (municipal disposal units) as well as informal/rural communities. Factors such as low operational costs, zero/low energy input during the composting period, potential for labour-intensive operations and reduced machinery requirements were considered in the selection of appropriate technologies. The results of the study pointed to the dome aeration technology (DAT) treatment in passively aerated windrows as a suitable option (Paar, 1999a; b; Mollekopf et al., 2002; Trois and Polster, 2006). Experience in Germany has shown that 'chimney' technology can achieve 90% reduction of landfill emissions, making it an appropriate technique for countries where pretreatment is still not a minimum requirement (Münnich et al., 2006).

A pilot project was designed to study the applicability of aerobic waste composting to the South African waste management context and prevailing subtropical conditions. This paper

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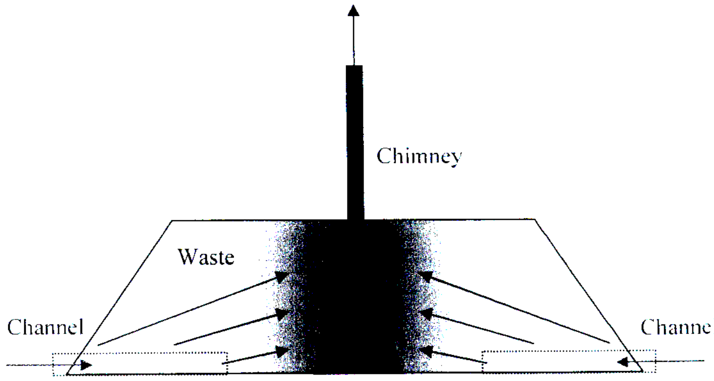
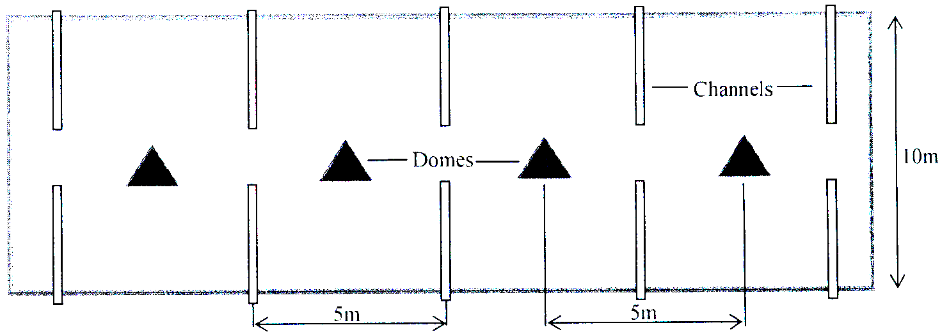
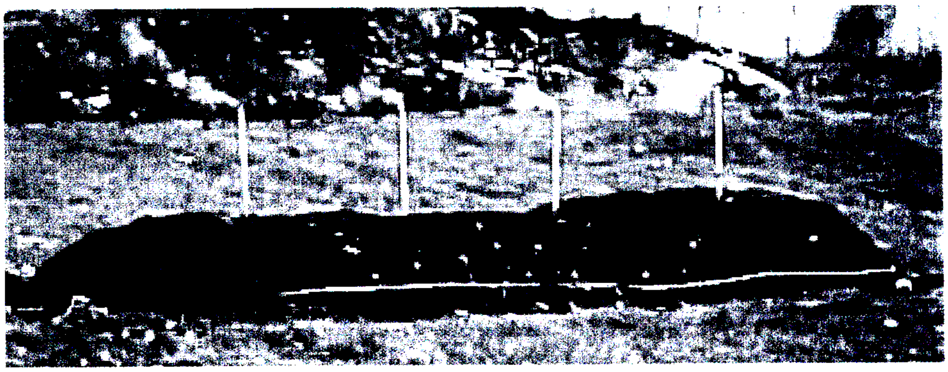
Figure 1 (top right)
The thermally driven aeration process of the DAT windrows

Figure 2 (middle right)
Schematic plan view of typical DAT windrow

Figure 3 (bottom right)
A completed DAT windrow during the early intensive degradation stage







describes the operational techniques adopted and focuses on the results of an investigation on the long-term impacts of the treated waste in anaerobic reactors (lysimeters) in relation to waste composition and rate of flushing. The pilot project of waste treatment and the lysimeter studies conducted in Durban are described in this paper.

Principles of the dome aeration technology (DAT)

Mechanical-biological treatment (MBT) is a two-stage process, involving the mechanical preparation of the waste, followed by biological composting. The mechanical treatment step removes useful fractions and prepares the biodegradable portion for optimal microbial activity. The process includes sorting, shredding/homogenisation and adjustment of moisture content. The most commonly implemented biological treatment stage is thermophilic aerobic composting in open systems or closed reactors (Soyez and Plickert, 2002). Composting reactors are closed vessels that utilise mixing or a forced air supply to ensure aeration. The advantage of these systems is their high efficiency and control over odour and leachate, although their complexity makes this technology expensive (Münnich et al., 2006). Open systems or non-reactor systems use windrows of waste which are aerated either by turning, through forced aeration or by natural thermal convection (passive aeration). The open composting systems are generally more cost-effective; however, there is less control over the process and thus degradation and emission control are less efficient than in reactor systems (Haug, 1993; Stegmann, 2005).

DAT is a passive aeration system that utilises thermal advection caused by temperature differences between the degrading material and the outside environment to drive the aeration process within a windrow of waste (Mollekopf et al., 2002). Steel mesh structures with a PVC chimney (called domes) which are placed in the centre of the windrow create a large air vent for hot gases flowing out of the system, and colder ambient air is drawn into the waste through lateral base channels (Fig. 1). Domes and channels are arranged in a staggered layout in order to maximise the distribution of air through the waste (Fig. 2). A completed windrow is shown in Fig. 3.

A high porosity of the waste is required for effective air-flow and is achieved by mixing the general waste with a more rigid structural material (SM) generally comprised of woody waste and dry branches from garden refuse. The site operations entail a three-step mechanical pretreatment stage of mixing, crushing and wetting, followed by composting for three to four months (Paar et al., 1999a; b, Mollekopf et al., 2002; Trois et al., 2005; Trois and Polster, 2006). After the composting is complete the material passes through a 60 to100mm screen. The fine fraction may be used as cover material for other windrows while the rest is landfilled.

Application of DAT in South Africa: Durban case study

Three full-scale DAT windrows were constructed at the Bisasar Road Landfill site and monitored for 20 weeks. Size and layout

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TABLE 1 Summary of the construction parameters for the Durban trials			
	Windrow 1	**Windrow 2**	**Windrow 3**
Waste input (Ratio)	50% Solid waste, structural waste (1:1) 50% pine bark	Solid waste, structural waste, pine bark (2:1:1)	Solid waste, structural waste, pine bark (2:1:1)
Machinery usage	Landfill compactor, front end loader, water truck, excavator	ADT, water truck, excavator	Front-end loader, ADT, water truck, excavator
Number of domes	3 Domes waste 3 Domes bark	3 Domes of waste	4 Domes of waste
Windrow size	325 m^3 waste 325 m^3 bark	350 m^3	450 m^3
Time required to construct	20 h	20 h	12 h

of the windrows were modified from the German experience to take into account the influence of local climatic conditions, different nature and composition of the waste, available construction and operational facilities. Samples of waste were collected before treatment and after 8 and 20 weeks of treatment for characterisation and for input into the lysimeters.

Climatic conditions

Since the DAT flow mechanisms rely on the thermal gradient between the ambient air and the gas generated within the windrow, it was assumed that higher ambient temperatures could reduce the aeration rates. Durban's climate is typically subtropical: warm and humid with a mean annual temperature of 20.5°C and average annual precipitation around 1 000 mm (concentrated between October and April). Daily minimum temperature averages between 21°C in summer and 10°C in winter. Maximum temperatures, on the other hand, average around 17°C and 30°C in winter and summer, respectively. The relative humidity is very high and often exceeds 70% in the summer. As the system is reliant on the thermal gradient between the windrow and the ambient air, concerns were raised over the effect of the higher temperatures experienced in Durban when compared to Germany. As a result, a conservative approach was adopted, so that the Durban windrows were reduced in size to 30 m long, 10 m wide and 2.5 m high (20% smaller than the German windrows (Trois et al. 2005)) as a smaller windrow would require less efficient aeration. In order to cover the entire seasonal spectrum the 1st windrow was constructed in June, the 2nd in December and the 3rd in April.

Construction and operation of the windrows

Construction and operation techniques adopted for the Durban trials were based on the experience of the Cottbus Landfill in Germany, where post-treatment of residues from high-tech MBT (with refractory-organic fine fraction remaining) is achieved to EU standards in only 3 to 4 months, with no turning (Mollekopf et al., 2002). The material received at general waste landfills in Durban is primarily constituted by 46% municipal solid waste, 40% inert material, 7% builders rubble and 7% garden refuse by mass (Bisasar Rd weighbridge data 2002-2005 – source Durban Solid Waste). The large percentage of inert material and builders' rubble is used for daily cover for the waste as stipulated by DWAF (1998). Approximately two thirds of the municipal solid waste (MSW) is degradable (putrescibles, paper and cardboard) while the remainder comprises plastics, glass and metals (DMWS, 1998). MSW from Rotor-press trucks was used as the rotary compaction shredded the waste.

It must be noted that there is no separate collection conducted apart from garden refuse which is also disposed of at MSW landfill sites in a different waste stream. In this investigation pine bark was also used to supplement the SM and as biofilter cover material to maintain the initial moisture content at optimal levels (55 to 65%) throughout the duration of the treatment. A summary of the windrow construction parameters is presented in Table 1.

Maximum temperatures reached indicate that 70°C is the upper limit for thermophilic composting using DAT. However, a problem was encountered in Windrow 2 where the biological activity was drastically reduced due to desiccation of the waste. Total CO_2 production indicated that Windrow 2 was biologically active for an effective time of only 2 to 8 weeks, although the target period was 20 weeks of composting. Thus, although studies were conducted on this waste, it was considered to be unrepresentative of 20 weeks pretreated waste and so the results are not presented herein. Windrow 3 remained biologically active for the entire composting time and the pretreatment was considered to be a success. A more detailed description of the windrow trials and results of the process monitoring phase are presented in Trois et al. (2005).

Laboratory tests in anaerobic reactors were carried out to simulate the long-term behaviour of the treated waste once landfilled. Three insulated 1 000 ℓ low density polyethylene tanks were installed with a leachate recirculation system consisting of a stone drainage layer, a pump, sprinkler and two sampling points – one at the base and one in the centre of the waste. Waste temperature and biogas measurements were conducted weekly through sampling points at the top of the tank. Each lysimeter was filled with approximately 500 ℓ of un-compacted waste as follows: Lysimeter 1 with fresh (un-composted) waste from construction of the third windrow (control reactor) and Lysimeter 2 with 8 weeks pretreated waste from a section of the third windrow. The input material to the lysimeters was characterised using eluate/leaching tests on coarsely shredded waste samples. After 24 h leaching (TS to liquid ratio of 1:10) the eluates were coarsely filtered (Filter paper type Whatmann 40) and tested for TS, VS, COD, BOD_5, $N\text{-}NH_3$, N-NOx, TKN and TOC using standard procedures (ASTM, 2004). Moisture content, total and volatile solids were determined on the solid matter (ASTM, 2004). The waste was weighed before placement in the lysimeters and the moisture content raised to field capacity (Table 2). Field capacity is the moisture content at which waste no longer retains additional water, a requirement if liquid samples were to be extracted. Note that only average values of minimum five repetitions are reported in Table 2; accuracy checks conducted weekly on the measurements confirm that the overall error was maintained at between 5 to 10%.

TABLE 2 Characterisation of the input material to the lysimeters				
Parameter	Unit	Un-treated waste	8 Weeks MBT	EU require-ments
Solids test results				
Moisture content	% Total mass	49	38	-
Volatile solids	% Dry mass	70	44	-
Field capacity	% Total mass	59	54	-
Eluate test results				
pH		7.05	6.83	5.5-13.0
COD	mg/ℓ	13090	2850	-
BOD_5	mg/ℓ	2272	443	-
$N\text{-}NH_4$	mg/ℓ	60.17	13.56	<200
N-NOx	mg/ℓ	50.8	5.44	-
TOC	mg/ℓ	3181	831	<250

The lysimeters were operated as unsaturated mixed reactors using closed leachate circulation, as implemented in similar investigations (Stegmann, 1982; Leikam et al., 1997; Höring et al., 1999). It must be noted that recirculation of leachate differs from conditions at full-scale landfill sites in South Africa where this practice is not applied. Recirculation, in lysimeter studies, serves to increase the contact between waste, water micro-organisms and nutrients (Kylefors et al, 2003). Thus the effective waste degradation is expected to be higher than that of full a scale landfill where preferential flow paths and areas of reduced permeability exclude areas of the waste body from contact with percolating leachate. Once a week, 2.5 ℓ of leachate were sampled from the bottom of the lysimeters and tested according to standard methods (ASTM, 2004) for the following parameters: Conductivity, pH, alkalinity, BOD_5, COD, $N\text{-}NH_3$, TKN, N-NOx. Levels of CH_4, CO_2 and O_2 (% by volume) were measured weekly using an infrared gas analyser (Model GA94) together with internal temperature readings using a thermocouple.

Due to the slowing down of biological activity over time, the release of pollutants from a landfill site in the long term is primarily reliant on physical leaching. The degree of leaching for a particular body of waste is described using the liquid to solid ratio (L/S) which represents the volume of water that has passed through the landfill body and is expressed as volume per unit mass of waste (dry mass). The assumption is that in the long term the physical leaching is not time dependent, but rather reliant on the flux of water through the waste. This allows for a direct comparison with full-scale landfill sites (Leikam et al., 1997; Kylefors et al. 2003; Robinson et al., 2005b). The testing period extended for 273d and, based on a weekly extraction of 2.5 ℓ, resulted in a cumulative L/S ratio of 0.55. This represents approximately 30 years of infiltration for a landfill 20 m deep with a dry density of 600 kg/m³ and an annual infiltration rate of 250 ℓ /m²·a (Leikam et al., 1997). The 250 ℓ/m² is a generalised figure as the landfill water balance is site-specific (infiltration at the Bisasar Road Landfill site is approximately 170 ℓ/m²·a). This is based on the assumption that the waste is at field capacity and the leachate released is equivalent to the annual infiltration. The results for significant leachate quality parameters are presented below in relation to the cumulative L/S ratio used in this investigation. The evolution of L/S with time is presented in Fig. 4.

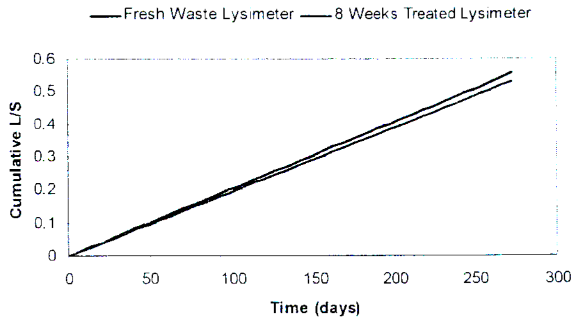

Figure 4
Evolution of L/S for both lysimeters with time

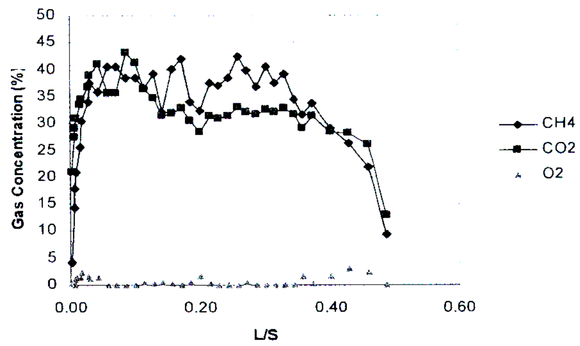

Figure 5
Biogas concentrations in Lysimeter 1

Results and discussion

Lysimeter 1: Fresh untreated waste

Biogas emissions from Lysimeter 1 are shown in Fig. 5. The methane concentration increased to 40 to 45% immediately after commencement of the trials. This rapid increase indicates that there was a limited acidic inhibition during the early stages of degradation. The period after the onset of methanogenesis is generally the stage of peak biological activity (Robinson, 1989). Although biogas volumes were not measured, the decrease in biogas concentrations towards the end of the test is indicative of slower production kinetics as ambient air begins to diffuse into the lysimeters headspace back through the biogas vent. This is evident in the increase of nitrogen gas concentration (not shown here), which accounts for the balance of the gasses in the lysimeters. The oxygen levels remain low due to consumption by facultative anaerobic and aerobic micro-organisms in the upper layers.

Figure 6 shows the evolution of COD and pH in the leachate from Lysimeter 1. Following an initial increase to 5 500 mg/ℓ, the organic content in the leachate falls to steady levels of approximately 600 mg/ℓ. The BOD to COD ratio at the final stages of the study was 0.4 indicating that a relatively large proportion of degradable organics was still present. At an L/S = 0.2 (after 100 d), the leachate is typical of a methanogenic state, with low COD concentrations and a neutral pH. The rapid decrease in COD is indicative of the activity of the methanogenic bacteria, as the intermediate products of anaerobic digestion are converted into methane and carbon dioxide.

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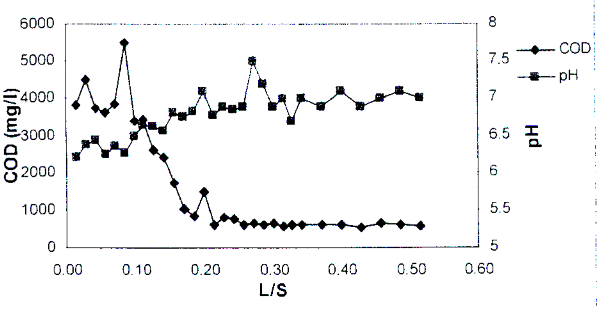

Figure 6
Evolution of pH and COD in the leachate from Lysimeter 1

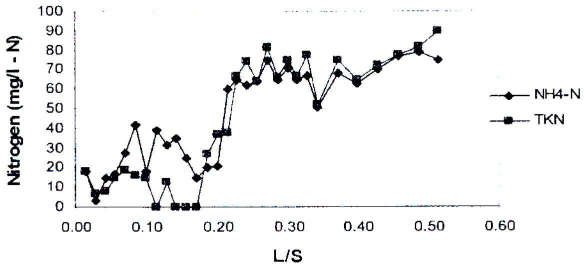

Figure 7
Evolution of NH_4-N and TKN in the leachate from Lysimeter 1

The evolution of the nitrogen compounds is shown in Fig. 7. The total Kjeldahl nitrogen (TKN) rises steadily as proteinaceous molecules are broken down by the microbial activity and nitrogenous compounds released into the leachate. The majority of the TKN is in the form of ammoniacal nitrogen which reaches steady levels of approximately 80 mg/ℓ at the end of the analysis period. Nitrates and nitrites were detected in negligible concentrations (generally <0.1 mg/ℓ) indicating that any aerobic activity within the lysimeter was insignificant. An unusual occurrence was the measurement of TKN levels lower than that of the ammoniacal nitrogen. Such readings, also reported by Bone et al. (2003) are physically impossible as TKN is the sum of ammoniacal and organic nitrogen and cannot be explained.

The COD and NH_4-N did not reach high levels as in other investigations (Novella et al., 2001). Furthermore, there was a negligible acidic inhibition evident in the early stages of the degradation process, as was found in other studies on untreated waste (Novella et al., 1997; Leikam et al., 1997; Cossu et al., 2003). This may be attributed to the large proportion of slowly degradable structural material (woody waste and pine bark) present in the lysimeter. The decrease in biogas (CH_4 and CO_2) concentrations coupled with the stabilisation of ammonia-nitrogen levels indicates that the rate of biological activity slowed towards the end of the trials. However, the relatively high BOD to COD ratio (0.4) suggests that a significant portion of the leachate was organic and that biodegradation was still taking place as a result of the slowly degradable fractions (BOD: COD ratios of 0.7 or greater are found in organic rich leachates (Robinson, 1989) while ratios of less than 0.1 are expected for leachates from highly degraded waste (Robinson et al., 2005).

Lysimeter 2: 8 weeks treated waste

Biogas emissions from Lysimeter 1 are shown in Fig. 8. The biogas readings show a rapid increase in CO_2 levels as residual

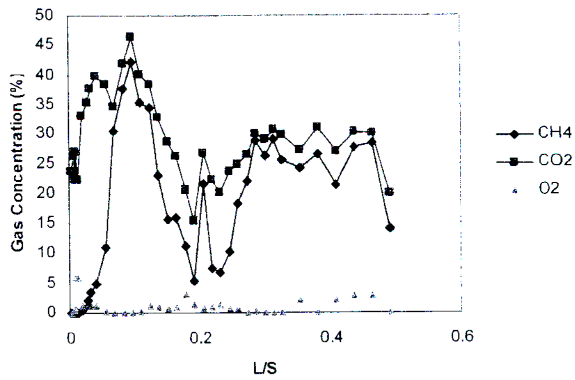

Figure 8
Biogas concentrations in Lysimeter 2

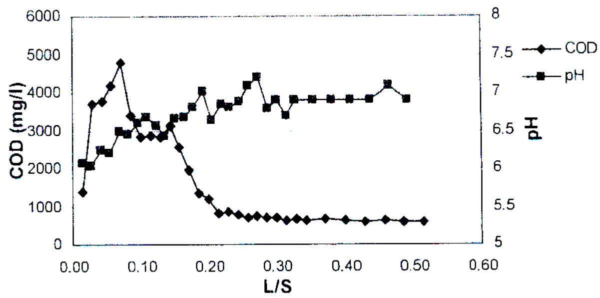

Figure 9
Evolution of pH and COD in the leachate from Lysimeter 2

atmospheric oxygen is consumed by aerobic bacteria. After the oxygen has been exhausted the methanogenic bacteria become established. It must be noted that the biogas concentration readings were affected by a break in the seal of the lysimeter lid. Once the broken seal had been discovered, the lid was resealed and the gas levels returned to expected levels.

A typical pattern of anaerobic degradation is evident in Fig. 9, with an early peak of COD which then decreases as the methanogenic bacteria become established. There is no acidic inhibition of the methanogenic activity, although the biodegradation kinetics are slow at the start of the test (shown by the slow uptake of oxygen) indicating low quantities of readily degradable organic compounds. Methanogenesis is achieved at L/S = 0.2 (after 100 d) indicated by the significant drop decrease in COD levels and a neutral pH. The drop in biogas concentrations towards the end of the testing indicates a decrease in biological activity. The methanogenic COD levels remain steady at approximately 600 mg/ℓ with a COD: BOD ratio of 0.4. As for Lysimeter 1, this indicates that a significant proportion of degradable organics were still present in the leachate at the end of the test period.

The TKN, most of which is in the form of ammoniacal nitrogen, increases steadily to 22 mg/ℓ (Fig. 10). The ammoniacal nitrogen concentration at the end of the test was 16 mg/ℓ – far lower than that of the raw waste in Lysimeter 1. These low levels of ammoniacal nitrogen are a feature of aerobically pretreated wastes (Robinson et al., 2005). As with Lysimeter 1, the TKN was at times measured to be lower than that of the ammoniacal nitrogen.

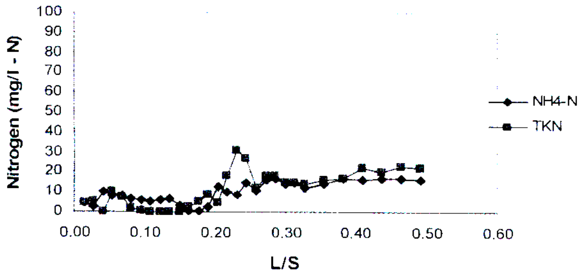

Figure 10
Evolution of NH_4-N and TKN in the leachate from Lysimeter 2

Comparisons

The onset of methanogenesis occurs within a similar period of time for both lysimeters, despite varying organic loadings. Both lysimeters showed no sign of acidic inhibition during the establishment of a methanogenic microflora. The rapid decrease in COD concentrations demonstrates the effectiveness of the methanogenic micro-organisms in the removal of the residual organic compounds. If acidic inhibition can be avoided, then the anaerobic microbial activity can be utilised as an effective biological treatment after landfilling. This would offer some degree of flexibility to account for areas in the windrows that are not properly aerated and thus require further treatment.

The stable methanogenic leachates from both lysimeters show similar COD and BOD levels. The concentration of non-biodegradable COD is equal for both lysimeters. As the material for the lysimeters was from the same source, this suggests that these compounds would not have been affected during the aerobic treatment stage. A significant benefit of waste pretreatment is evident in the levels of ammoniacal nitrogen, which are substantially higher for the fresh waste than the pretreated waste.

The dependence of the DAT system on a significant proportion of woody waste (used for structural material) may detract from its suitability as it has been suggested that the slow degradation of these refractory organic materials may persist for decades (Robinson, 2000). The hypothesis that the long-term COD release is due to the breakdown of slowly degradable organic material such as wood and paper implies that these fractions should not be landfilled but rather subjected to further treatment.

Implications for long-term pollution prediction

The lysimeter results demonstrated the presence of a 'hard' or non-degradable COD, as described by Robinson et al. (2005). The leachate is also characterised by slowly biodegradable organics. These compounds are likely to present a persistent pollution problem. The current DWAF discharge standards were used as criterion for acceptability of the leachate discharge in natural receptors (DWAF, 1998b). The refractory COD concentration at the end of the testing is calculated by subtracting the biodegradable content (BOD) from the total COD levels (Table 3). This parameter is assumed to be biologically non-degradable and therefore removed only by physical leaching. Similarly, as the biological consumption of NH_4-N only occurs under aerobic conditions, its removal is also assumed to be reliant on physical processes. Further production of both refractory COD and ammoniacal nitrogen are assumed to be negligible. If the lysimeter is assumed to be a completely mixed reactor, the concentration of a pollutant (c) can be calculated after a given number of sampling cycles with Eq. (1), based on the Landsim Declining Source Term Model (Robinson, 2005b).

$$c = c_0 \exp\left(\frac{-V_s}{V_t}\theta\right) \tag{1}$$

where:

c is the pollutant concentration
c_0 is the initial concentration
V_s is the volume of leachate removed for sampling (replaced by distilled water),
V_T is the total lysimeter leachate volume
θ represents the number of sample cycles.

The number of sampling cycles is then used to determine the cumulative L/S at that point. Based on the weekly removal of 2.5 ℓ of leachate from the lysimeters, the L/S ratio required in order to meet discharge standards and the projection of the L/S ratio to a full scale landfill, as described above, are given in Table 3. The full-scale prediction is based on a 20 m deep landfill with an infiltration rate of 250 ℓ /m²·a and a dry density of 600 kg/m³ (Leikam et al., 1997). It must be remembered that the accuracy of the predictions is limited due to the heterogenic nature of landfill sites and differences between the experimental set up and a full scale site such as leachate recirculation and rate of leachate release (Kylefors et al, 2003). The extrapolation of the L/S ratio gives an idea of the timeframes that may be expected. Furthermore, landfills are assumed to be at field capacity, application of the approximations to sites in arid regions where leachate generation is sporadic is questionable although mathematically there is no distinction between continual flow or discreet leachate release, in the case of this particular model.

From the prediction of the time required to reach suitable discharge standards for the full-scale landfill the following comments can be made:

TABLE 2
Characterisation of the input material to the lysimeters

Parameter	Unit	Fresh waste	8 Weeks treated
COD			
Value at the end of trials	mg/ℓ	370	380
Discharge value (target)	mg/ℓ	75	75
Long-term L/S		2.8	2.4
Time required to reach target (Full-scale site)	years	140	122
NH_4-N			
Value at the end of trials	mg/ℓ	75	16
Discharge value (target)	mg/ℓ	15	15
Long-term L/S		2.8	0.7
Time required to reach target (Full-scale site)	years	141	29

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- The long-term COD release from landfills containing 8 weeks treated waste would be very similar to those containing untreated waste, as expected.
- The reduction of ammoniacal nitrogen levels in pretreated waste is significant after only 8 weeks of treatment.

The accelerated leaching out of pollutants may be possible by taking advantage of the high rainfall of the Durban area to increase the rate of liquid flux through the site to greater than 250 $\ell/m^2 \cdot a$. Another consideration is of the target concentrations which may be unreasonably stringent and research is underway to assess the nature of leachates from treated waste and assess the suitability of the current discharge standards.

Conclusion

The benefits of pretreatment are realised in reduced levels of ammoniacal nitrogen in the leachates. However, the organic loading in the methanogenic leachate of treated waste remains unacceptably high. Unless studies show that the leachates from pretreated waste should be subject to less stringent discharge standards, MBT alone is not sufficient to result in the cessation of landfill aftercare (leachate treatment) within sustainable timeframes.

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Long-term emissions from mechanically biologically treated waste: Influence on leachate quality – Part II

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Abstract

Mechanical biological pretreatment of waste prior to disposal is proven to effectively reduce the long-term polluting potentials of landfilled waste. The combined effect of waste pretreatment and flushing, as is possible in landfills operated in tropical or sub-tropical countries, has the potential to further reduce the landfills' environmental impact. In this study, long-term emissions from pretreated waste were monitored in anaerobic leaching columns operated at increasing liquid-to-solid ratios. The efficiency of the pretreatment, conducted in full-scale passively aerated windrows, was assessed by comparing different treatment periods (8 and 16 weeks). In order to understand the influence of sorting (separated collection) on the pretreatment, the treated waste was sieved in a 50mm diameter sieve and the coarse and fine fractions separately analysed in the leaching columns. The results showed that treating the waste markedly reduces the COD and NH_3-N loadings while the coarse fractions show a greater long-term pollutant risk.

Keywords: mechanical biological waste treatment, flushing, leaching columns, bioreactor landfill, leachate

Introduction

The increase in environmental awareness and the growing focus on sustainable development are changing the way in which modern engineers deal with solid waste disposal. In South Africa, the current approach is to 'concentrate and contain' and in the case of municipal solid waste (MSW), this includes entombing a large volume of degradable organic material in sanitary landfills that may produce highly polluted leachates. Due to the long timeframes required to reach stabilisation of the waste body, the focus has shifted towards treating the problem at the source, rather than dealing with the emissions of untreated waste (Robinson, 2000; Cossu et al., 2003).

Waste pretreatment prior to disposal is gaining momentum internationally as a possible solution. *The European Council Landfill Directives 1999/31/EEC* (LFD) require member states to only landfill wastes that have been subjected to prior treatment (Robinson et al., 2005). Mechanical biological pretreatment (MBP), in particular, has proven to reduce the organic loading in the leachate. The effectiveness of aerobic pretreatment on the removal of long-term ammonia loadings is still not clear.

The University of KwaZulu-Natal, in collaboration with Durban's Waste Disposal Unit (Durban Solid Waste – DSW), has conducted research on the behaviour of landfill emissions under a subtropical climate since 2000. It now appears evident that high rainfalls, typical of a subtropical climate, are favourable in promoting an optimum environment for biodegradation (Bowers, 2002). In 2002, the first South African pilot project on aerobic pretreatment of waste in passively aerated windrows was initiated at the Bisasar Road Landfill site in Durban. This note reports on an aspect of this study that investigates the possibility of coupling aerobic waste pretreatment with flushing in a bioreactor landfill in order to shorten the acetogenic stage and actively remove pollutants from the leachate. Note that a detailed description of the methodology followed in the windrows' construction and operation, including the assessment of the treatment performance and the preliminary results of the pilot project, was presented in Griffith and Trois, 2006. In this study, long-term emissions from pretreated municipal solid waste were monitored in anaerobic leaching columns operated at increasing liquid-to-solid ratios. The efficiency of the pretreatment was assessed by comparing different treatment periods (8 and 16 weeks). In order to understand the influence of sorting (separated collection) on the pretreatment efficiency and the contribution of the fine fractions in the overall organic loading in the leachate, the treated waste was sieved in a 50 mm diameter sieve and the coarse and fine fractions separately analyzed in the leaching columns.

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Materials and methods

The Dome Aeration Technology (DAT) was used for the treatment of municipal solid waste (MSW) in passively aerated open windrows set up at the Bisasar Road Landfill site in Durban (Paar et al., 1999; Mollekopf et al., 2002; Griffith and Trois, 2006; Trois and Polster, 2007; Trois et al., 2007). The pretreatment stage involved the mixing of MSW with bulky waste, comprised mostly of dry garden refuse, to maintain the high porosity required for effective aeration. The material was wetted before placement in DAT windrows which were 10 m wide, 30 m long and 2.5 m in height, in order to ensure 55 to 60% moisture content for optimum microbial activity (Trois et al., 2007). The waste was then aerobically composted for 8 and 16 weeks, as discussed in detail in Griffith and Trois, 2006.

Sieving the waste is employed in the MBP process both before and after the biological treatment stage in order to separate the high calorific value coarse fraction (size >40 mm) which is usually incinerated from the fine highly biodegradable material (size <40 mm) (Soyez et al., 2002; Kuehle-Weidemeier et al., 2003). A screen size of 40 to 100 mm is typically used; the material retained in the sieve (called upper-sieved) is generally incinerated, while the passing (called under-sieved) undergoes biological stabilisation. In this study, the screening was applied for a dual reason: to understand the effect that a preliminary sorting of recyclable materials (coarse fractions) from putrescibles (fine fractions) may have in the overall treatment efficiency and in the quality of the long-term emissions from landfilled treated waste. Waste was sieved in a rotating drum screen with an aperture size of 50 mm. The ratio of the upper-sieved (coarse) to the under-sieved (fines) was approximately 1:2.5 on a dry-weight basis. However, it was noted that the coarse material still contained a portion of fine particles that adhered to its surface and were not removed during the screening. The results of the physical characterisation conducted on the two fractions are presented in Table 1.

The waste samples derived from the windrows and untreated general waste collected from the Bisasar Road landfill site were characterised using standard tests on the solid material and 24 h leaching tests. Analyses on the solid matter included moisture content tests, total and volatile solids and static respiration index on 7 d (RI_7); 24h eluate tests were conducted using a liquid- to-solid ratio L/S = 10, i.e. 200g of solid is soaked in 2 000 mℓ of distilled water in an Erlenmeyer flask that is agitated for 24 h, and the eluate produced is then filtered in a 0.45 µm filter paper and analyzed for pH, conductivity, COD, BOD, NH_3, NOx, TS and VS. The tests were conducted for a minimum of 3 repeats with analytical methods in accordance with the *US Standard Methods* (2004).

TABLE 1
Composition of the treated and untreated waste samples

Component (% mass)	Untreated waste	Treated waste global	Treated waste fine	Treated waste coarse
Plastics	31	14.6	2.5	26.6
Fines	22	31	54.5	7.6
Paper	25	13.1	6.3	19.7
Fabric	4	13.6	1.2	25.3
Glass	7	4.2	7.7	0.8
Metal	6	2.8	0.8	4.6
Wood	0	3.9	0	7.4
Rubber	5	3.5	0	3.8
Stones	0	7.7	11.5	4.2
Plant matter	0	15.1	15.1	0
Bone	0	0	0.3	0

The different component-percentages of the waste samples tested are presented in Figs. 1, 2, 3 and 4 (Marchetti, 2007).

A limitation in the eluate tests is the short timeframe of the test which does not allow for full biochemical solubilisation.

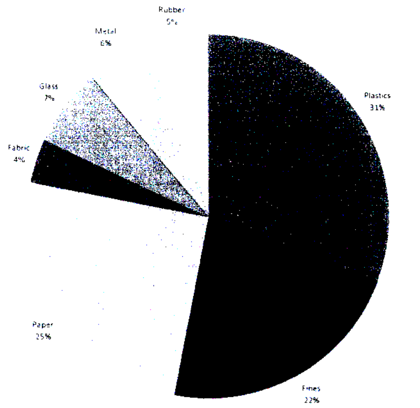

Figure 1
Composition of the untreated (fresh) waste sample

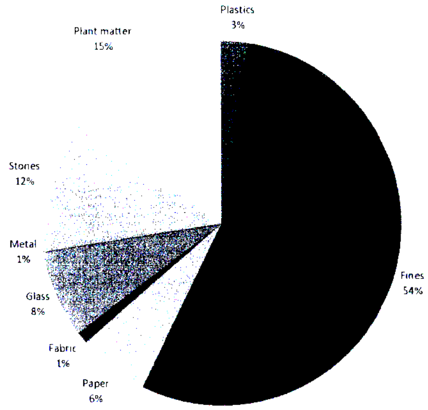

Figure 3
Composition of the lower-sieved fraction of the treated waste sample

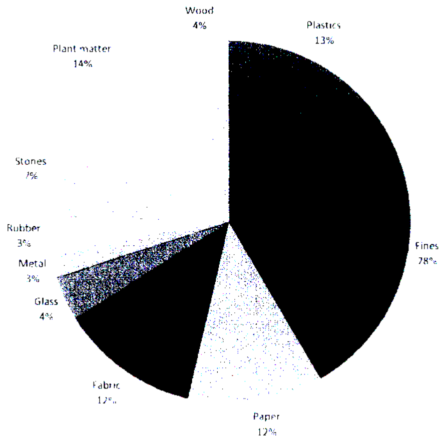

Figure 2
Composition of the treated global waste sample (before screening)

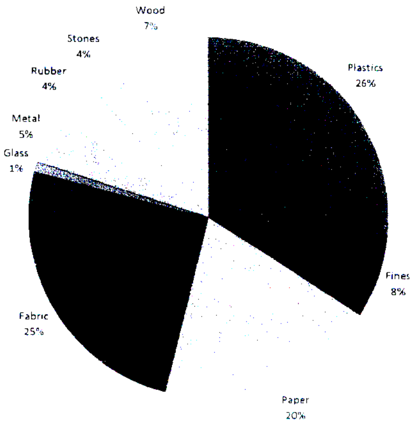

Figure 4
Composition of the upper-sieved fraction of the treated waste sample

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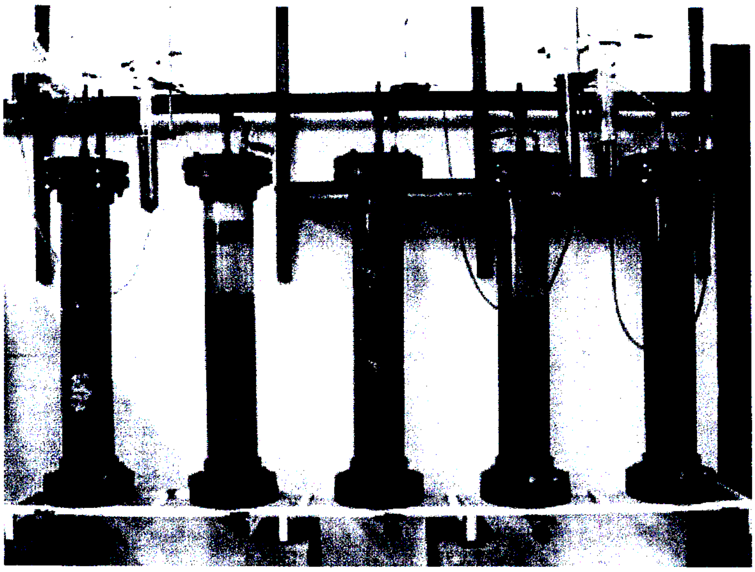

Figure 5

From left to right: Column 1: untreated waste; Column 2: under-sieved treated for 16 weeks; Column 3: over-sieved treated for 16 weeks; Column 4: under-sieved treated for 8 weeks; Column 5: over-sieved treated for 8 weeks

TABLE 2
Column configuration

Column	Type of waste	Duration of treatment	Particle size	Sample denomination
Column 1	Untreated MSW	-	-	Control
Column 2	Treated MSW	16 weeks	<50 mm	Under-sieved
Column 3	Treated MSW	16 weeks	>50 mm	Upper-sieved
Column 4	Treated MSW	8 weeks	<50 mm	Under-sieved
Column 5	Treated MSW	8 weeks	>50 mm	Upper-sieved

Thus leaching column tests were conducted in order to determine the leachate characteristics over a longer time frame.

The columns consisted of transparent PVC pipes, 160 mm in diameter and 1 m in length with a volume of 20 ℓ (Fig. 5). Approximately 15 ℓ of waste sample was weighed and inserted in the column with light compaction. The waste was then saturated by filling with distilled water to the 15 ℓ mark, with the remaining 5 ℓ serving as a headspace for gas analysis. During the tests, Columns 2 and 4 containing the fine fractions settled substantially while the coarse materials in Columns 1, 3 and 5 floated due to gas production (Fig. 5). Although the top level of the material deviated from the 15 ℓ mark, the volume of gas, liquid and solid remained constant.

On a weekly basis the columns were drained and refilled with deoxygenated distilled water. This *modus operandi* ensured that the columns were always in contact with water, the gas volume remained constant and that the column would behave as a mixed reactor. Extracting a smaller sample of leachate may have resulted in a plug flow scenario, with the leachate removed from the bottom not representative of the column leachate as a whole. A limitation in this method is that due to the varying densities of the material, the dry masses and consequently the liquid-to-solid ratios were different for each column. Thus direct comparison of the concentrations of the parameters tested is not possible and the mass of material solubilised between each sampling step must be assessed. Table 3 shows the mass of the input materials and volumes of water added as well as the weekly flux of water.

Each column was equipped with a gas monitoring set-up employing the liquid displacement method. The biogas quality was tested weekly using an infrared gas analyzer (Geotechnical Instruments – GA 2000) which provided percentages of carbon dioxide, methane and oxygen (volume/volume in air). The leachate extracted from the columns was analysed weekly for pH, conductivity, COD, BOD, TS, VS, NH_4-N and NO_x-N. The tests were conducted in a minimum of three repeats following analytical methods in accordance with the *US Standard Methods* (2004) for characterisation of water and wastewater. The columns' operation was ceased when the concentrations of parameters such as NH_3-N and COD fell below reliable analysis range.

Column processes

In order to properly assess the evolution of the leachate quality from the columns at increasing liquid-to-solid ratios, solubilisation patterns must be analysed. The following four primary processes occur during the course of the column operation:

TABLE 3
Column input and operation parameters

Parameter	Column 1	Column 2	Column 3	Column 4	Column 5
Waste type	**Untreated**	**16 weeks <50 mm**	**16 weeks >50 mm**	**8 weeks <50 mm**	**8 weeks <50 mm**
Mass (kg)	4.95	5.33	1.62	6.19	2.64
Initial water input (ℓ)	9.57	10.14	12.17	11.83	10.37
Weekly water output (ℓ)	2.56	4.78	7.93	5.66	5.94
Weekly L/S flux	0.52	0.90	4.91	0.91	2.25
Operation time (wks)	61	28	28	31	31
Final cumul. L/S ratio	39	26	95	29	65

- Solubilisation of solids into the liquid state through dissolution (Diss) and
- Hydrolysis (Hyd)
- Physical removal through the liquid flux (Phys)
- Biological conversion to biogas which is vented (Bio).

Thus the mass balance of a particular measured parameter between sampling can be expressed as follows:

$$C_{i+1}*Vc = C_i*Vc + Diss + Hyd - Phys - Bio \qquad (1)$$

where:

C_i is the concentration at week i

C_{i+1} is the concentration the following week

Vc is the volume of liquid in the column

As the volume of leachate sample removed (Vls) is known, the total mass physically removed (*Phys*) during the sampling exercise can be calculated:

$$Phys = C_i*Vls \qquad (2)$$

Thus Eq. (1) can be rewritten as follows:

$$C_{i+1}*Vc = Ci*Vc - Ci*Vls + Diss + Hyd - Bio \qquad (3)$$

From this formula, C_{i+1}, C_i, Vc and Vls are all known while *Diss*, *Hyd* and *Bio* are unknown. For simplicity, these unknowns are grouped into the single parameter Ψ, therefore:

$$Diss + Hyd - Bio = \Psi \qquad (4)$$

Rearranging and grouping similar terms in Eq. (3) and dividing by the dry mass of the solid material (M_{TS}), Eq. (3) becomes Eq. (5):

$$\psi = \frac{C_{i+1} \cdot V_C}{M_{TS}} - \frac{C_i(V_C - V_{ls})}{M_{TS}} \qquad (5)$$

Thus in the context of these laboratory tests, where the liquid flux is controlled, it is possible to quantify the reactions' Ψ. A positive Ψ indicates a net solubilisation of solids into the liquid phase with a negative Ψ showing a net conversion of dissolved compounds into biogas. The interpretation of this is that a material with a higher positive rate of solubilisation will cause a greater leachate load, and *vice versa*. The assumption in this assessment is that the solubilisation of the solids into the liquid phase is not significantly limited by their concentration.

Results and discussion

Eluate tests

The results of the 24 h eluate tests are shown in Table 4, as an average value of three repeats and the relative standard deviation.

The untreated waste shows significantly higher values for all pollutants as well as acidic pH. The difference between the upper-sieved and the under-sieved is less distinct and the comparable COD and volatile solids levels suggest that some organic fine particles remained attached to the upper-sieved during sieving.

TABLE 4
Chemical waste characterisation

Parameter	Untreated	8 Weeks		16 Weeks	
		Fine	Coarse	Fine	Coarse
pH	5.3±0.2	7.3±0.1	6.9±0.1	7.3±0.1	7.0±0.1
Conductivity (mS/cm)	6.2±0.01	1.53±0.16	1.71±0.2	1.41	1.84
COD (mg/ℓ)	7 598±131	1 489±484	1 475±216	3 161±304	3 640±360
NH_3-N (mg/ℓ)	48.5±30.5	14.63±0.14	10.43±0.14	27.23±0.01	23.3±0.01
NOx-N (mg/ℓ)	18.1±8.9	8.61±2.1	8.12±1.6	4.93± 0.01	5.9±0.01
TS (g/ℓ)	15.62±0.73	2.64±0.79	2.2±0.26	7.31±0.01	5.12±0.01
VS (g/ℓ)	7.13±0.29	1.13±0.41	0.67±0.09	2.62±0.01	2.8±0.01

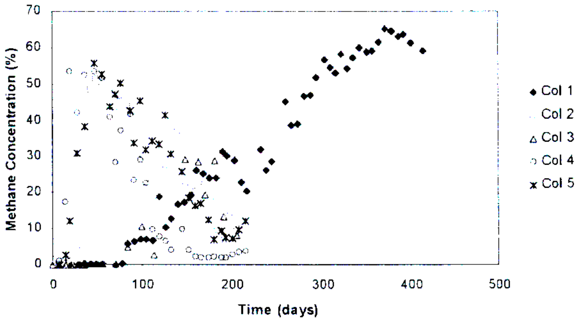

Figure 6
Evolution of methane concentrations with time in the columns

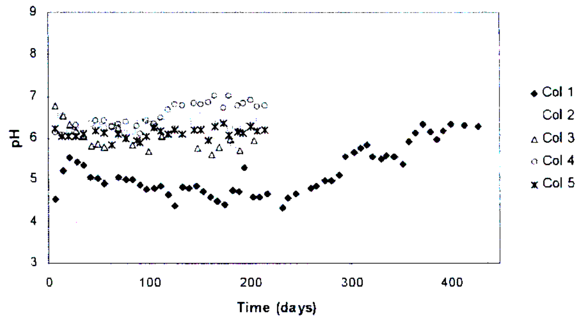

Figure 7
Evolution of pH with time in the effluent from the columns

Column tests – Biogas and pH

The evolution with time of the methane concentrations from the columns and the pH in the leachate are presented in Figs. 6 and 7.

Column 1 experienced acidic methanogenic inhibition typical of untreated waste, with no methane generation in an acid environment at first, followed by a gradual increase for both parameters. The methane concentrations of Columns 2, 4 and 5 rise rapidly soon after commencement of the tests showing no signs of acidic inhibition, a result confirmed by the leachate pH which remains above 6 for both treated fractions, with particular evidence for the under-sieved Columns 2 and 4. Methane production was initially inhibited in Column 3, due to the high porosity of the over-sieved that favoured semi-aerobic conditions before reaching methanogenesis that was achieved more rapidly by saturating the airspace above the solid waste with nitrogen gas during each irrigation, thus also reducing the negative effect of large head-spaces formed after the settlement of the fine material (Columns 2 and 4).

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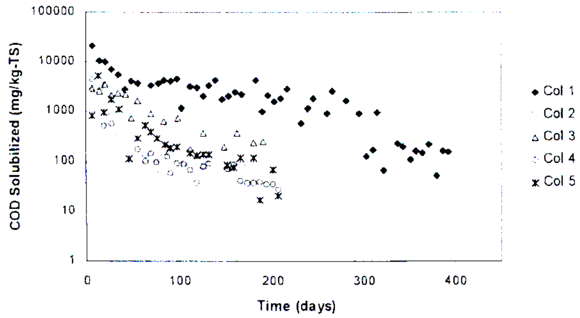

Figure 8
Evolution of COD loadings from the columns

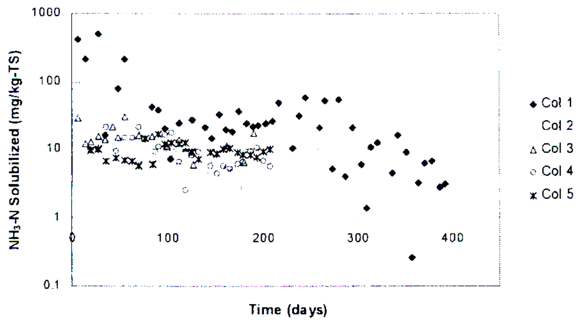

Figure 9
Evolution of NH_3-N loadings from the columns

Leaching processes

The degradation processes occurring in the leaching columns were described by monitoring indicator parameters COD and ammonia. The evolution of these indicators over time in the effluents from the columns is reported in Figs. 8 and 9.

The results in Fig. 8 enable comparison of organic removal achieved with and without pretreatment, showing the benefit of the pretreatment in effectively reducing the COD concentrations in the leachate. This benefit is particularly evident for the under-sieved treated material in Columns 2 and 4, if compared with the untreated global sample in Column 1. This seems to confirm the initial hypothesis that the under-sieved material contains more readily degradable fractions and that the over-sieved fraction is constituted primarily by slowly or non-degradable matter (Figs. 1 to 4). The almost comparable results between the behaviour of the over-sieved and the under-sieved, particularly before reaching methanogenic conditions, may be explained by an inefficient sieving technique that allowed organic fine fractions to be trapped onto the surface of the coarse material. It is interesting to note the similar behaviour for different durations of pretreatment, which indicates inefficiencies in the treatment of the 16 weeks' waste, as discussed in Griffith and Trois (2006).

Figure 9 shows the solubilisation of ammoniacal-nitrogen during the leaching process. The untreated waste (Column 1) initially releases significantly more ammoniacal nitrogen than the treated waste, with differences becoming less apparent with time. The difference between the NH_3-N levels for the different treated wastes is less marked.

The cumulative pollutant load at the end of the test was calculated and a mathematical extrapolation of the solubilisation patterns was used to project the ultimate pollutant release from each column, as described under '**Column processes**' above. These results are compared to the eluate test results and are presented in Table 5.

TABLE 5
Cumulative COD and NH_3-N loads

Waste type	Units	Cumulative COD	Cumulative NH_3-N
Untreated	mg/kg-TS		
Eluate		75 987	485
Column 1		15 8254	2 097
16 Weeks' Fine			
Eluate		34 230	11.1
Column 2		26 933	1 002
16 Weeks' Coarse			
Eluate		34 570	9.5
Column 3		34 481	428
8 Weeks' Fine			
Eluate		12 430	14.6
Column 4		13 936	338
8 Weeks' Coarse			
Eluate		17 390	10.4
Column 5		29 563	485

The comparison between the eluate tests and the leaching columns show how the duration of the test affects the results, with higher cumulative COD loads recorded from the materials containing the slowly degradable fractions while the results from the fine material are similar to the eluate test results. The difference is more significant when considering the NH_3-N results, with the longer time-frame allowing for hydrolysis of the proteinaceous material and thus a higher cumulative load of NH_3-N is determined from the columns.

The eluate tests show clearly the benefit of a mechanical biological treatment prior to disposal, with a significantly lower pollutant loading measured in the treated material as compared with the untreated control. However, the small difference between the coarse and fine fractions of treated waste observed after 24 h of eluate tests suggests an inefficient sieving (mechanical) treatment.

Nonetheless, Table 1 and Figs. 1 to 4 show that screening the waste is an effective means of separating non-degradable (plastics) and slowly degradable (paper) recyclable materials from the waste stream. Difference in size is marked with the majority of the fine material being smaller than 10mm, while the majority of the coarse material is significantly greater than 50 mm. The potential use of the fine material as daily cover should be further investigated.

The leaching tests in columns show that the untreated waste carries a far greater pollutant load in both the short and long term than the treated fractions. Acidic inhibition was observed in the test on untreated waste, but was absent for all the treated samples. Furthermore, the untreated waste exhibits a slower leaching rate and thus presents a longer risk to the environment. The comparison between coarse and fine material shows that although the under-sieved carries a higher leaching potential in the early stages, it is the over-sieved fraction that presents a higher risk in the long term due to the slower decrease in solubilisation.

The results also confirm that the 16 weeks' treatment was less efficient than the 8 weeks' treatment to fully stabilise the waste, with the material from the latter displaying a lower pollutant loading from the eluate tests. The efficiency of the treatment process in open windrows decreases with time by increasing its sensitivity to negative conditions such as desiccation, and temperature variations as presented in Griffith and Trois (2006).

Conclusions

The biological aerobic treatment of MSW is effective in reducing the pollutant load of the material, with significant reductions in COD and NH_3-N loading. The screening of the waste was successful in separating the fine fractions from what is primarily paper and plastic – materials that could be removed from the waste stream due to their potential for recycling. These fractions are also of higher calorific value and may be of benefit if waste-to-energy projects are to be considered. Furthermore, the benefit of source separation lies not only in savings of landfill airspace, but also in the reduction of long-term leachate loadings after landfilling.

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