UNIVERSITY OF SOUTHAMPTON

FACULTY OF ENGINEERING AND THE ENVIRONMENT

School of Civil Engineering and the Environment

Biodegradation and Settlement Behaviour of Mechanically Biologically Treated (MBT) Waste

by

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Thesis for the degree of Doctor of Philosophy

April 2011

UNIVERSITY OF SOUTHAMPTON

ABSTRACT

FACULTY OF ENGINEERING AND THE ENVIRONMENT

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Doctor of Philosophy

BIODEGRADATION AND SETTLEMENT BEHAVIOUR OF MECHANICALLY BIOLOGICALLY TREATED (MBT) WASTE

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Mechanical biological pretreatment processes are increasingly being employed as a means of diverting biodegradable municipal waste from landfill to comply with the EU Landfill Directive. The long term behaviour of mechanically biologically treated (MBT) waste will be different from that of unprocessed municipal solid waste (MSW) since the pretreatment process may change its physical, chemical and biological properties.

The aim of this research was to investigate the biodegradation and settlement behaviour of MBT waste. Large scale laboratory experiments using consolidating anaerobic reactors (CARs) have been carried out on MBT wastes, treated to typical UK and German standards. Gas generating potential, leachate quality, settlement characteristics and hydraulic properties of MBT wastes were determined and compared with raw MSW. A detailed characterisation of the MBT waste and its associated chemical and physical properties was a key component of the study. The stabilisation of MBT waste was achieved in less than a year under enhanced biodegradation conditions. The research has demonstrated the benefits of pretreatment in:

-substantially reducing the gas generating potential

-releasing low levels of total organic carbon, ammoniacal nitrogen and heavy metal contents in the leachate

-reducing the long term settlements due to creep and biodegradation

The settlement of the waste was divided into immediate compression, primary settlement (consolidation), and secondary settlement. The contributions of mechanical creep and biodegradation to secondary settlement were identified separately and mechanical creep was found to be the more significant of the two. The secondary settlement data were analysed using simple settlement models and the model parameters were found to be suitable for the estimation of long term settlement. The overall trends and results of hydraulic conductivity and drainable porosity against density and stress for the pre-treated waste were similar to those for the raw MSW.

Twelve small scale BMP reactors were run in parallel with the CARs to characterise the anaerobic biodegradability of the MBT waste in terms of biogas potential and solids composition. The solids composition, leachate parameters and biogas yield were measured at various stages of degradation through the sacrifice of reactors at various times. The biogas potential was shown to correlate well with cellulose plus hemicellulose to lignin ratio ((C+H)/L), loss on ignition and total carbon content of the waste indicating a clear link between these parameters.

Irrespective of whether it is MBT waste or raw MSW, the relationships of hydraulic conductivity against density, biodegradation induced settlement against gas produced, and change in (C+H)/L ratio versus biogas potential are all similar and they all plot on the same line.

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DECLARATION OF AUTHORSHIP

I, Asif Ali Siddiqui, declare that the thesis entitled Biodegradation and Settlement Behaviour of Mechanically Biologically Treated (MBT) Waste and the work presented in the thesis are both my own, and have been generated by me as the result of my own original research. I confirm that:

- this work was done wholly or mainly while in candidature for a research degree at this University;
- where any part of this thesis has previously been submitted for a degree or any other qualification at this University or any other institution, this has been clearly stated;
- where I have consulted the published work of others, this is always clearly attributed;
- where I have quoted from the work of others, the source is always given. With the exception of such quotations, this thesis is entirely my own work;
- I have acknowledged all main sources of help;
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- parts of this work have been published as:

Siddiqui A. A., Richards D. J., and Powrie W. (2009). A preliminary analysis of mechanically biologically treated waste: biodegradation and settlement behaviour. Proceedings of Sardinia (2009), Twelfth International Waste Management and Landfill Symposium, S. Margherita di Pula, Cagliari, Italy; 5 - 9 October 2009.

Siddiqui A. A., Powrie W., and Richards D. J. (2010). Long-term biodegradation and settlement behaviour of mechanically biologically treated waste: laboratory study. Proceedings of WASTE 2010:Waste and Resource Management- Putting Strategy into Practice, Stratford-upon-Avon, Warwickshire, England, 28-29 September 2010.

Siddiqui A. A., Powrie W., and Richards D. J. (2011). Experimental study on anaerobic degradation of MBT waste. Proceedings of The 26th International Conference on Solid Waste Technology and Management , Philadelphia, USA, 27-30 March 2011.

Siddiqui A. A., Powrie W., and Richards D. J. (2011). Experimental study on anaerobic degradation of mechanically biologically treated waste. Fourth International Workshop on the Hydro-Physico-Mechanics of Landfills (HPM4), Santander, Spain; 27-28 April 2011.

Siddiqui A. A., Powrie W., and Richards D. J. (2011). Investigations on landfill behaviour of pre-treated waste. To be presented at the Thirteenth International Waste Management and Landfill Symposium to be held in Sardinia , Italy from 3-7 October 2011.

Signed: ……**Date:** …….

Acknowledgements

The PhD research is a long journey with its alternating good and bad moments, but everyday we learn something in either the professional or personal sense. I can say now that the journey had a positive balance and the experience was great: I learnt a lot and made good friends from all around the world. But to be honest, some times in this trip I felt as if I were sitting in a boat with a hole in the bottom, in the middle of the ocean. Fortunately in these moments I had my family and friends who helped me to drain the water, fix the boat and arrive safely at the objective.

I am very grateful to my supervisors Prof. William Powrie and Prof. David Richards for their intellectual guidance, competent advice, encouragement and continuous support throughout the research period. Completion of the work required for this degree would have been impossible without them. I have learnt a great deal from them during the past three and a half years at the University of Southampton.

I would also like to thank Dr. Richard Beaven and Dr. David Smallman for their valuable comments and pertinent suggestions for this research work.

Both technicians in the laboratory, Mr. Harvey Skinner and Ms. Pilar Pascual deserve my great thanks for their assistance in setting up the experimental apparatus, solving the instrumental problems and helping with my analytical work.

I am grateful to all my friends for their support and encouragement. The opportunity to undertake this research at the University of Southampton was possible through the funding of EPSRC.

I am indebted to my family members and relatives for their encouragement and for daring me to take on the challenge of this PhD study which had been in my mind for the past decade.

Certainly the biggest thank is due to my wife, for her patience and support during this nerve-racking period, and my daughter, whose lovable and innocent faces, gestures and talk brushed off all the cumulated fatigue of the usually tiring day.

ABBREVIATIONS AND ACRONYMS

ADF	Acid detergent fibre
ADL	Acid digestible lignin
BMP	Biochemical methane potential
BMW	Biodegradable municipal waste
BS	British standard
CARs	Consolidating anaerobic reactors
COD	Chemical oxygen demand
(C+H)/L	Cellulose plus hemicellulose to lignin ratio
DM	Dry matter
DOC	Dissolved organic carbon
EC	Electrical conductivity
EDWS	European drinking water standards
EU	European Union
GB21	Gas formation potential after 21 days
HBM	Hydro-bio-mechanical
IC	Inorganic carbon
LDAT	Landfill degradation and transport
L/S	Liquid to solid ratio
LOI	Loss on ignition
MBP	Mechanical biological pretreatment
MBT	Mechanically biologically treated
MSW	Municipal solid waste
NDF	Neutral detergent fibre
NH_4-N	Ammoniacal nitrogen
PSD	Particle size distribution
STP	Standard temperature and pressure
TC	Total carbon
TN	Total nitrogen
TOC	Total organic carbon
VFA	Volatile fatty acids
VS	Volatile solids

LIST OF NOTATION

α_c	Creep parameter
$C_{\alpha\varepsilon}$	Coefficient of secondary settlement
c_v	Coefficient of consolidation
d	Maximum drainage path length
E'_o	Constrained modulus
$\Delta\sigma'_v$	Increment of vertical stress
Δh	Change in waste sample height
h	Waste height, different subscripts are used for different stages of settlement
ε_c	Creep induced settlement strain
ε_b	Biodegradation induced settlement strain
ε_{bt}	Total settlement strain resulting from biodegradation
ε_{total}	Total secondary settlement strain
k	Saturated hydraulic conductivity
k_b	Degradation rate constant
ρ_d	Dry density
ρ	Bulk density
n	Total porosity
n_d	Drainable porosity
T	Time factor, dimensionless parameter (Section 5.1.2)
V_d	Volume of drainable voids
V_v	Volume of voids
V_s	Volume of waste solids

CHAPTER 1 INTRODUCTION

1.1 Background

Landfill has been the dominant municipal solid waste (MSW) management option for disposal of residual wastes in the UK and many other countries for over a century. The long term pollution potential and ongoing waste settlement are among the principle concerns for landfill management. Continued biodegradation after landfilling leads to biogas and leachate production, and settlements occur both during the operating period and for a long time after landfill closure. Settlements are important because of the need to be able to estimate the remaining void space to achieve the required fill levels, while post closure settlements can adversely impact the integrity of the capping system.

The EU Landfill Directive (EC, 1999) sets targets for all Member States to reduce substantially the amount of biodegradable MSW going to landfill. The Directive requires a stepwise reduction in the amount of biodegradable municipal waste (BMW) being disposed of to landfills in the UK to 75% of 1995 levels by 2010, 50% by 2013 and 35% by 2020. Many European countries (e.g. Germany, Austria) required to comply with the Directive earlier than the UK have chosen mechanical-biological pretreatment (MBP) as a technology for treating MSW to arrive at the targets set out in the Landfill Directive. MBP normally involves sorting to remove recyclables and, in some cases, combustible materials; particle size reduction (e.g. shredding and screening) and partial biodegradation by anaerobic digestion and/or aerobic

composting processes. A large number of new waste treatment facilities have been commissioned over recent years and the role of MBP in waste management is expected to become more popular in the short term.

The focus of the Landfill Directive is solely mechanistic and no official limits or evaluation criteria were specified for the biodegradability parameters of the final product which is intended for landfilling. In Germany, waste pretreatment processes are more advanced and a Landfill Ordinance (German EPA, 2001) has set very strict allocation criteria for the landfilling of MBT waste to ensure high standards in terms of several parameters e.g. the TOC content of the waste and the biogas production of waste over 21 days under anaerobic conditions measured in a GB21 test. In contrast, the UK has not defined standards for landfilled MBT waste and a quantitative approach is employed to reduce the amount of BMW which is landfilled in accordance with the Directive. A BM100 test, which measures the biogas production over 100 days, is carried out before and after the treatment and the percentage reduction in biogas produced is taken as reflecting the percentage reduction in BMW after treatment. Therefore, mechanically biologically treated (MBT) wastes in the UK are likely to be of a lower standard (i.e., not as well sorted or processed) than in Germany. Germany has already met the Landfill Directive's 2016 target to landfill not more than 35% of the amount of BMW generated in 1995 (Herczeg and Reichel, 2009).

Numerous studies (de Araujo Morais et al., 2008; Bockries et al., 2003; Leikam and Stegmann, 1997) have focused on the difficulty of reaching the stability criteria defined in Germany. Although, MBT wastes treated to German standards are less biologically active, they still degrade, produce gas

and leachate, and settle over a long period of time (Zach et al., 2000; von Felde and Doedens, 1999; Ziehmann & Meier, 1999; Leikam et al., 1999); these problems will be worse with waste that has been treated to a lesser extent, as in the UK.

Pretreatment will have major implications on the degradation and settlement characteristics of the waste in landfills. The long term behaviour of MBT waste will be different from that of unprocessed MSW since the pretreatment process may change its physical, chemical and biological properties. Landfilling of MBT waste in the UK and other parts of Europe is likely to continue into the foreseeable future. Therefore, an understanding of the gas generating potential, leaching behaviour and settlement characteristics of MBT wastes (resulting from a variety of standards of treatment) has long term relevance.

Knowledge of the gas generating potential and leaching behaviour of MBT wastes will improve on ability to assess the potential risks posed by these landfills to the environment. The gassing potential and rate are important parameters in sizing the gas collection and control system. Knowledge of the leaching behaviour helps in designing a leachate treatment facility. Settlement is a vital element of effective landfill design, and post closure settlement may damage the cover and gas or leachate collection systems. Long term settlements in landfill may arise as a result of mechanical creep and biodegradation. These two mechanisms are distinct and must be considered separately though their effects in terms of settlement against time are quite similar (Powrie et al., 2009). Therefore, an approach is needed to predict the long term settlement.

MBP plants have been in operation for many years in other countries outside the UK. However, MBT waste studies are at an early stage and further research is required. Data and experience on the performance of MBT landfills i.e. those filled exclusively with MBT, is not currently available. Robinson et al. (2005) investigated the impact of biological pretreatment on leachate quality. Limited data are available on the long term leachate quality and gas generating potential of MBT waste, based on small scale studies (e.g. Bayard et al., 2008; Bockreis and Steinberg, 2005; Horing et al., 1999; Leikam and Stegmann, 1999). These studies demonstrate that MBT wastes have reduced gas generating potential and leachate strength. Unfortunately, studies about the release of heavy metals in leachate from MBT waste are scarce. Mass balances provide one way of checking the reliability of data. Surprisingly, this is seldom done, most likely because gathering the data to conduct these balances may necessitate additional sampling and monitoring of the experimental system. Mass balance calculations require rigorous linked monitoring of input and output waste, and gaseous and liquid phases.

None of the studies on MBT waste to date have included consideration of the settlement characteristics, and therefore uncertainties remain about the creep and biodegradation induced settlements. Biogas and leachate characteristics have been reported by a few authors. However, no studies have captured the complete stabilisation process including settlement characteristics. Moreover, there is no information on the characterisation of anaerobic biodegradability in terms of the solids composition change with the progression of decomposition. Therefore, biodegradation and settlement behaviour of MBT waste need to be quantified and linked. Moreover, a clear distinction between settlement due to mechanical creep and biodegradation needs to be

made. This comprehensive study represents perhaps the most complete set of data available to date on settlement, biogas production, leachate quality and anaerobic biodegradability of MBT waste.

1.2 Aims and objectives

Experimental results for the long term biodegradation and settlement behaviour of the UK MBT and German MBT wastes are obtained, analysed and compared. The data are also compared with raw MSW study (Ivanova, 2007, 2008*a*, 2008*b*) previously carried out at the University of Southampton. This research is intended to contribute to knowledge and practice in attaining the following aims:

1. To investigate the effect of pretreatment on the gas generation potential, leaching behaviour, settlement and hydraulic conductivity of biodegradable municipal solid waste.

2. To investigate the impact of the level of pretreatment on gas generation potential, leaching behaviour, settlement and hydraulic conductivity of biodegradable municipal solid waste.

3. To investigate the suitability of simplified settlement models as an assessment tool.

These aims will be achieved by fulfilment of the following objectives:

- to obtain complete data sets in terms of biogas, leachate, settlement, and solids composition as the waste degrades

- to evaluate the stabilisation of the waste in terms of gas generation potential, leachate quality and settlement
- to characterise the settlement of waste due to mechanical creep and biodegradation
- to analyse settlement data in context of simplified settlement models
- to characterise the anaerobic biodegradability of the waste and establish a link between biogas potential and solids composition
- to carry out carbon and nitrogen mass balances to examine the transformation of carbon and nitrogen from solid to liquid and gaseous phases
- to study the changes in hydraulic properties of waste with compression

for two specimens of MBT waste, treated to typical UK and German standards.

CHAPTER 2 LITERATURE REVIEW

The landfilling of municipal solid waste (MSW) may pose threats to human health and the environment because of the production of the leachate and gas resulting from biodegradation of organics in the waste. Leachate containing organic and inorganic substances can contaminate the surrounding soil, groundwater and surface water, and the gas rich in methane contributes to the global warming if released to the atmosphere. Degradation leads to a reduction in the waste volume and waste settlements that are transferred to the capping system. The diversion target of the EU Landfill Directive is directed at the biodegradable part of MSW as this fraction causes significant problems. This is the reason for the development of mechanical biological pretreatment (MBP) in Western Europe as a main option to reach the EU Landfill Directive target.

In a typical mechanical biological pretreatment facility, the mechanical treatment sequence consists of a combination of sorting, separation, shredding and screening technologies with the purpose of maximising resource recovery and conditioning the waste for subsequent treatment. The biological treatment, aimed at reducing the biological activity of the waste, may include composting or anaerobic digestion or anaerobic digestion followed by composting. The final product after biological treatment is either a compost-like product or a stabilised biodegradable material. Therefore, MBT wastes have a smaller particle size, fewer biodegradable organics and a greater inert fraction than unprocessed municipal solid waste. MBP is

becoming increasingly popular as a method for treating MSW to fulfil the requirements of the EU Landfill Directive and this technology is expected to become more popular in the short term.

In Germany, the Waste Storage Ordinance for environmentally sound landfilling of MSW (AbfAbIV, 2001) defines criteria for the waste to be landfilled as shown in Table 2.1.

Table 2.1: Criteria for landfilling of MBT waste in Germany

Parameter	Permissible Limit
TOC_{solid}	≤ 18% dry mass
Gross calorific value	≤ 6000 KJ/kg
Gas formation rate, GB_{21}	≤ 20 litres/kg DM
TOC_{eluate}	≤ 250 mg/L
Ammoniacal nitrogen $_{eluate}$	≤ 200 mg/L

In contrast to Germany, there are no set standards for MBT wastes in the UK. The expected effects of MBP are the minimisation of leachate and biogas production, reduction of bad odours and reduction of landfill settlement, which in turn may reduce the landfill aftercare period (Robinson et al., 2005; Soyez and Plickert, 2002; Damiecki, 2002).

Landfill design, management and aftercare requires knowledge of the pollution potential and settlement characteristics of MBT wastes. Moreover, landfill owners and regulators have to consider leachate quality, gas generating potential and settlement for the long term management of a

landfill. Various researchers have examined the biodegradation and settlement behaviour of MSW. This will now be discussed as both a background and a benchmark against which to assess the biodegradation and settlement behaviour of MBT waste.

2.1 Anaerobic biodegradation of MSW in landfills

Landfills are very complex environments and sustain numerous microbiological and physicochemical reactions during sequential phases of stabilisation. The organic matter found in solid waste includes cellulose, hemicellulose and lignin. Barlaz et al. (2002) have shown that cellulose and hemicellulose (comprising 45 to 60% of MSW) are major biodegradable constituents responsible for 90% of methane production in the landfills. Biodegradation of solid waste and gas generation in landfills are governed by a series of chemical and biological reactions through which solid organic particles are solubilised and converted to biogas. The four important anaerobic degradation steps carried out by a consortia of anaerobic bacteria are:

Hydrolysis – solid and complex dissolved organic matters are broken down by hydrolytic bacteria into smaller soluble components required for subsequent microbial conversions.

Acid fermentation – fermentative bacteria converts dissolved organic matter (hydrolysis product) into volatile fatty acids (VFA), alcohols, hydrogen and carbon dioxide. This process results in high concentration of VFA.

Acetogenesis – acetogenic bacteria converts the longer chain VFA and alcohols into acetate.

Methanogenesis – methanogenic bacteria converts acetic acid into methane and carbon dioxide.

Landfills receiving MSW proceed through a series of four distinct phases viz. phase 1 (aerobic phase), phase 2 (anaerobic acid phase), phase 3 (accelerated methane production phase) and phase 4 (decelerated methane production phase) (Barlaz et al., 1989*b*). In the acidogenic phase, the concentration of VFA rises to a peak and the pH reaches its lowest. There is a concurrent increase in inorganic ions due to the leaching of easily soluble material in the acidic environment. Leachate generated at this stage is of high organic strength and the gas evolved comprises mainly carbon dioxide and hydrogen. The low pH increases the mobility and solubility of heavy metals and therefore high metal concentrations are observed in this phase. The methanogenic phase is characterised by a high pH (close to neutral) and steady methane production, a low concentration of VFA and a decline in leachate strength.

In practice, the identification of acidogenic and methanogenic conditions has been considered important as these phases seem to have markedly different leachate (Kjeldsen et al., 2002; Robinson et al., 2005) and gas characteristics (Bockreis & Steinberg, 2005; Reinhart & Al Yousfi, 1996; Barlaz et al., 1990). The methanogenic phase promotes the immobilization of heavy metal ions by facilitating the formation of metal hydroxides, sulphides, carbonates and complexes with organic material. Cellulose and hemicellulose decomposition begin in phase 3 and continue in phase 4. As cellulose and hemicellulose are surrounded by lignin, they will not be fully decomposed due to the inhibitory effect of lignin. During waste decomposition, the cellulose and

hemicellulose content decreases while the lignin content increases (i.e. it forms a greater proportion of the remaining waste), as lignin is highly recalcitrant and stable under anaerobic conditions.

The stabilisation of MSW is one of the main indicators relating to the evaluation of the long term emission potential of the landfills. Research conducted during the last three decades has identified the key process parameters that influence waste stabilisation process in the landfills. The degradable organic fraction in the waste, moisture content, pH and temperature of the system appear to be the most important. However, moderate to severe inhibitions can be caused by heavy metals and trace organic compounds in the leachate if present above certain concentrations (Reinhart and Townsend, 1998; Kjeldsen et al., 2002). Leachate recirculation is the most-used enhancement technique for waste stabilisation by increasing the moisture content, redistributing the microorganisms and nutrients within the waste matrix and diluting the high concentrations of inhibiting substances. Addition of sewage sludge has a positive impact on the stabilisation process (Reinhart and Townsend, 1998; Knox, 2000).

There has been an increased emphasis on the operation of landfills as bioreactors in the past decade to enhance the degradation process and accelerate waste stabilisation, i.e. to bring the landfill to an inert state in a relatively short time (Benson et al., 2007; Sponza and Agdag, 2004, Warith, 2002 Reinhart et al., 2002). Bioreactor landfills have been suggested as a more sustainable alternative to conventional landfilling. Researchers have outlined the benefits such as increased biogas production, lower leachate strength, increased settlement and shorter stabilisation period (Benson et al., 2007;

Sponza and Agdag, 2004; Reinhart and Townsend, 1998; Reinhart and AlYousfi, 1998).

After closure, landfills need to be managed and controlled to avoid adverse effects on humans and the environment. The aftercare period of a landfill can only be terminated if landfill does not pose a threat to the environment (IWM, 1999). There are no agreed standards against which a landfill can be described as having reached its final stabilisation. Many authors have tried to define a "stabilised" landfill waste and several research studies have shown that the leachate strength is a relevant indicator to determine the stabilisation of landfills (Francois et al., 2006; Berthe et al., 2007). Whereas, according to Borglin et al. (2004), the waste in landfill is considered stabilised when leachate is no longer a pollution hazard, gas production is negligible or stopped and the majority of settlement has occurred. A general opinion is that a landfill is monitored until it has reached a stable state characterised by the gas generation, leachate characteristics and settlement.

Francois et al. (2007) studied the impact of leachate recirculation on degradation of MSW and found the waste to be stabilised within 400 days. Valencia et al. (2009) conducted pilot scale experiments on MSW to achieve final storage quality status and showed that the bioreactor landfill was capable of achieving practical stabilisation after 2 years of operation.

2.2 Biochemical methane potential and biodegradability studies

The biochemical methane potential (BMP) test measures the methane or biogas that can potentially be produced under anaerobic conditions by a known quantity of waste. This test is well recognised to provide a reliable estimate of organic waste biodegradability (Godley et al., 2005; Wagland et al., 2009; Hansen et al., 2004; Eleazer et al., 1997). The basic principle of tests to estimate biodegradability is to assess how much of the carbon can be mineralised and how quickly it will be degraded. BMP tests are bioassays in which a sample is incubated in a temperature controlled system, with nutrients and bacteria added to optimise conditions for microbial methanogenesis.

There are several batch anaerobic digestion methods for measuring the BMP of waste with the basic approach of incubation of sample with nutrients and bacteria under optimised microbial methanogenic conditions (usually mesophilic at 30°C). However, the technical approaches in terms of particle size of the sample, inoculum, gas measurement technique and duration of the test vary significantly among the published methods, which depend on the purpose of the study and the type of waste sample measured (Hansen et al., 2004; Eleazer et al.,1997; Owen and Chynoweth, 1993; Owen and Stuckey, 1979). In general the larger the sample, the more representative that sample is of the waste material, and so the more reliable and valid the data.

Table 2.2: Cellulose plus hemicellulose to lignin ratio for raw MSW and decomposed waste

Reference	*Raw MSW*	*Partially degraded waste*	*Well decomposed waste*
Bookter and Ham[a] (1982)	4.0	0.9-1.2	0.2
Hossain et al. (2003)	2.52	-	0.25
Wang et al. (1994)	-	-	0.016-0.21
Zheng et al. (2007)	2.0	1.1	-
Ivanova et al. (2008*b*)	3.24		0.43

Note: [a] represents cellulose to lignin ratio

There has been a wide range of studies on the degradability of MSW buried in landfills. The relative concentrations of cellulose (C), hemicellulose (H) and lignin (L) have previously been used to assess the degree of decomposition of landfilled waste at various stages (Stinson and Ham, 2005; Bookter & Ham, 1982; Wang et al., 1994; Baldwin et al., 1998). As shown in Table 2.2, the ratio of C/L or (C+H)/L has been used as a measure of refuse composition and degradability, and typically decreases with increasing waste decomposition (Wang et al., 1994; Hossain et al., 2003; Mehta et al., 2002).

Bookter & Ham (1982) analysed shredded MSW samples in lysimeters and reported that the C/L ratio of fresh refuse was about 4, of partially decomposed refuse 0.9 - 1.2 and of well decomposed refuse about 0.2. Wang et al. (1994) reported (C+H)/L ratios in the range 0.016 - 0.21 for well decomposed samples from old landfills which is quite a wide range. They

couldn't find any relationship between (C+H)/L ratio and BMP for the decomposed samples. Eleazer et al. (1997) characterised the anaerobic biodegradability of different components of MSW (e.g. grass, leaves, branches, food and paper) by measuring methane yield and the extent of cellulose and hemicellulose decomposition. It was found that 81% of cellulose and 58% of hemicellulose were degraded after 135 days. Barlaz et al. (1997) found that methane yield increases with increase in cellulose and hemicellulose content. These studies on the degradability are generally consistent in demonstrating that the (C+H)/L ratio decreases with the decomposition of MSW. While the relationship between methane production and solids decomposition is well understood, there have apparently been few if any attempts to link biogas potential with (C+H)/L ratio in a formal way. To my knowledge there is no other literature that relates (C+H)/L ratio to biogas potential.

Baldwin et al. (1998) analysed samples from a MSW landfill for cellulose, hemicellulose and lignin over a period of 6 years and found that materials with a high initial lignin content degraded very little whereas those with little or no lignin showed very high losses in cellulose and hemicellulose. Hossain et al. (2003) quantified the state of waste decomposition using the (C+H)/L ratio, which decreased from 2.52 to 0.25 as the waste degraded over a period of 127 days. Zheng et al. (2007) showed a decrease in (C+H)/L ratio from 2.0 to 1.1 for MSW composted partially over a period of 120 days. A correlation was also shown between biogas yield and (C+H)/L ratio.

In summary, research has documented the depletion of cellulose and hemicellulose, and relative enrichment in lignin with degradation of MSW.

Previous research has not captured the biodegradability of MBT waste, and there is a lack of information on solids composition change with degradation of an MBT waste. The current research provides new knowledge in this field by characterizing the anaerobic biodegradability of MBT waste using BMP, total carbon (TC), loss on ignition (LOI), cellulose, hemicellulose and lignin contents. The research will also define relationships between biogas potential and solids composition.

2.3 Previous studies on biodegradation of MSW

A number of authors have studied the evolution of leachate quality and gas generating potential of MSW. Leachate constituents of concern for the long term pollution potential of landfills are total organic carbon (TOC), ammoniacal nitrogen (NH_4-N) and heavy metals (Kjeldsen et al., 2002; Price et al., 2003; El Fadel et al., 2002). These constituents are released either by direct leaching or during degradation of organic and nitrogenous compounds. They may cause environmental problems e.g. toxicity, oxygen demand etc. if the leachate migrates into surface water or groundwater.

2.3.1 Field scale studies

Benson et al. (2007) studied the effect of leachate recirculation in five full-scale landfills in North America. The characteristics of leachate were studied from year 1992 to 2002 in terms of pH, chemical oxygen demand (COD) and ammoniacal nitrogen concentrations. The results indicated that the landfill reached methanogenic stage in the first 2 years after leachate recirculation and pH increased and remained around 7 and 8. They found that although initial COD values in excess of 10,000 mg/litre had fallen to below 1000

mg/litre within four years, concentrations of ammoniacal nitrogen showed little change, remaining at about 600-800 mg/litre throughout the period.

The long term performance of full scale MSW landfill facilities with leachate recirculation was monitored by Morris et al. (2003). These landfills are located in the Central Solid Waste Management Centre, Delaware, USA. They reported an increase in pH of the leachate from 5 to about 7 and decrease in biochemical oxygen demand (BOD) from 10,000 mg/litre to 100 mg/litre over a period of about seven years of leachate recirculation. The ammoniacal nitrogen concentration in the leachate also decreased slightly from 600 mg/litre to about 400 mg/litre for the period of study.

Yuen (1999) studied the leachate quality based on a two years study on landfill and reported ammoniacal nitrogen~ 500 mg/litre, TOC ~ 1000 mg/litre and chloride ~ 4000 mg/litre. Sormunen et al. (2008) studied leachate quality at a full scale MSW landfill for about 2 years and found ammoniacal nitrogen and chloride in the range of 500-2000 mg/L.

Existing data showed high leachate concentrations of all components in the early acidogenic phase due to strong decomposition and leaching. In the long methanogenic phase a more stable leachate with lower concentrations is observed. The changes observed in the quality of leachate from full scale landfills mainly cover the transition between the acidogenic degradation and the subsequent methanogenic degradation (Table 2.3).

The range of values cited in Table 2.3 is too large to be useful and demonstrates that there is no such thing as a typical leachate, but that it reflects the stage of the landfill and contents of the waste.

Table 2.3: Composition of acidogenic and methanogenic landfill leachate

Parameter[1]	Acidogenic phase			Methanogenic phase			Overall range (Kjeldsen et al. 2002)
	Average Germany (Ehrig 1989)	Average UK (Robinson and Gronow 1993)	Range[2]	Average Germany (Ehrig 1989)	Average UK (Robinson and Gronow 1993)	Range[2]	
pH	6.1	6.7	4.5 - 7.8	8	7.5	6.8 - 9	4.5 - 9
COD	22000	36817	6000 - 152000	3000	2307	500 - 8000	140 - 152000
TOC	-	12217	1010 - 29000	-	733	184 - 2270	30 - 29000
NH_4-N	750	-	30 - 3000	750	-	30 - 3000	50 - 2200
Total nitrogen	1250	-	50 - 5000	1250	-	50 - 5000	-
Chloride	2100	-	100 - 5000	2100	-	100 - 5000	150 - 4500
Calcium	1200	2241	10 - 6240	60	151	20 - 600	10 - 7200
Magnesium	470	384	25 - 1150	180	250	40 - 478	30 - 15000
Cadmium	0.006	0.02	-	0.006	0.015	-	0.0001 – 0.4
Copper	0.08	0.13	-	0.08	0.13	-	0.005 - 10
Chromium	0.30	0.13	-	0.3	0.09	-	0.02 – 1.5
Lead	0.09	0.28	-	0.09	0.20	-	0.001 - 5
Nickel	0.2	0.42	-	0.2	0.17	-	0.03 - 1000
Zinc	5	17	0.1 - 140	0.6	1.1	0.03 – 6.7	-

Notes: [1] All results are in mg/litre except pH value

[2] Based on data from Ehrig (1989) and Robinson and Gronow (1993)

2.3.2 Laboratory scale studies

The impact of leachate recirculation on degradation of MSW in simulated landfill bioreactors has been investigated by several researchers e.g. Francois et al. (2007), Erses et al. (2008), Bilgili et al. (2007), Sponza and Agdag (2004) and Warith (2002).

Kylefors et al. (2003) showed that long term prediction of leachate qualities are more appropriate using landfill simulator reactors compared to the small scale shaking leaching tests.

Erses et al. (2008) examined degradation of MSW for a period of 700 days and observed an acidogenic phase period of 290 days. They found an increase of TOC to 11000 mg/litre in the acidogenic phase which was then decreased to about 300 mg/litre in methanogenic phase. The NH_4-N concentration remained at about 1000 mg/litre throughout the study period. Methane yield was reported as 158 L/kg DM (litres per kilogram dry matter) with methane forming about 60% of the total biogas produced.

Warith (2002) carried out an experimental study to determine the effects of solid waste size, leachate recirculation and nutrient balance on the rate of MSW biodegradation. This study indicated that the smaller the size of the MSW the faster the biodegradation rate of the waste. The pH of the leachate was in the range of 7 to 8 after two years of leachate recirculation. A decreasing trend of the organic load, measured as COD and BOD, was observed. The concentration of chloride in the leachate remained fairly constant at about 1000 mg/litre during the entire period of study.

Francois et al. (2007) studied the impact of leachate recirculation on the degradation of MSW in bioreactors and shown that the waste stabilisation reached after 400 days of degradation. They also highlighted the accumulation of chloride ion and ammoniacal nitrogen due to the leachate recirculation.

Bilgili et al. (2007) examined the effect of leachate recirculation on anaerobic degradation of MSW in laboratory scale landfill reactors. The acidogenic phase was reported as 30 days and waste stabilisation was achieved after 500 days. Sponza and Agdag (2004) explained that leachate recirculation reduce stabilisation time, enhance methane gas production and improve leachate quality. Bilgili et al. (2007) and Sponza and Agdag (2004) noticed a decrease in ammoniacal nitrogen concentration in leachate over the period of their studies.

These small scale-scale studies have provided sufficient evidence to suggest that the concept of bioreactor landfills is technically viable. However, these small scale experiments have their own limitations. Although they can allow the flexibility to study a large number of operational variables under controlled conditions, it is obvious that they cannot accurately simulate the natural degradation processes in full-scale landfills due to the scale effects. For example, almost all of the small scale studies worked with shredded waste but very rarely the same treatment was given to the MSW in full-scale landfills. In addition, the kind of recirculation rate and uniformity of moisture distribution that can be achieved in a laboratory test cannot be obtained easily in a full-scale landfill cell. However, the research and development devoted to full-scale investigations are still relatively limited.

2.4 Previous studies on biodegradation of MBT waste

Horing et al. (1999) investigated the gas and leachate emissions from MBT waste in a laboratory scale study. They reported TOC, total nitrogen and chloride content of leachate in the range 0.3 - 3.3, 0.6 - 2.4 and 4 - 6 g/kg DM respectively. Leikam and Stegmann (1999) studied the behaviour of MBT waste in landfill simulation tests, in comparison with MSW. For the MBT waste, the acidogenic phase during which strong organic leachate is produced was absent, and after about 250 days the TOC of the leachate was below 400 mg/litre. A much more significant benefit of pre treatment becomes apparent when concentrations of total nitrogen (TN) are considered. Whereas the TN content in leachate from MSW stabilised at about 1000 mg/litre, this value was below 200 mg/litre for pre-treated waste. They reported a substantial reduction in the biogas potential of MBT waste compared to MSW, although the precise basis of this claim is uncertain.

Robinson et al. (2005) investigated the impact of biological pretreatment on leachate quality. They reported TOC and ammoniacal nitrogen (NH_4-N) concentration of leachate in the range 500 – 2000 mg/litre and 50 - 1000 mg/litre respectively based on the degree of composting. Table 2.4 presents range of leachate parameters for MBT wastes based on the degree of composting. Van Praagh et al. (2009) looked at heavy metals in pre-treated waste using batch leaching test, but the behaviour of heavy metals in terms of release potential via leachate pathway in landfills is still unknown. The leachate quality data available in literature do not allow a comparison on the basis of the leachate load (defined in g/kg DM) owing to an insufficiency of information.

Table 2.4: Composition of leachate from landfilled MBT waste (Robinson et al. 2005)

Parameter	Degree of composting	
	High	Low - Medium
pH	7.5 - 8	7.5 – 8.5
Conductivity (mS/cm)	6 - 10	10 - 20
COD	1000 - 1500	1000 - 5000
TOC	500	500 - 2000
NH_4-N	30 - 200	50 - 1000
Chloride	1000 - 2000	4000 - 8000
Calcium	250 - 300	100 - 800
Magnesium	60 - 100	100 - 400
Chromium	0.05 – 0.1	0.1 – 0.5
Nickel	0.1	0.1 – 0.7
Copper	0.2	0.2 – 0.5
Zinc	0.2 – 0.5	0.5 – 3.0
Cadmium	0.003	0.005 – 0.1
Lead	0.02 – 0.04	0.1 – 0.4

Note: All results in mg/litre except pH value, and conductivity (mS/cm)

Gas generating potentials of pre-treated wastes in the range 5 -50 litre/kg DM were reported in separate studies (Bockreis and Steinberg, 2005; De Gioannis et al., 2009, Horing et al. 1999). Binner and Zach (1999) assessed the biological stabilisation of MBT waste by investigating its gas generating potential in the laboratory and showed that MBT waste produces 95-100% of its gas generating potential within 240 days but this, in reality must depend on the operational circumstances. It would get more difficult to achieve with larger waste bodies. Nearly all of these studies have shown the absence of acidogenic phase. In contrast, an acidogenic phase lasting for a few months was demonstrated by Sormunen et al. (2008) for the initial stages of MBT waste lysimeters. Concerns have been expressed that although the

biodegradability would have been significantly reduced, thereby reducing the emission rate, the time scale of gas production may remain. A study by Knox and Robinson (2007) for example, found that gaseous emissions occurred over the same time period as for untreated wastes. Other studies (Komilis et al., 1999; Mahar et al., 2007) have found that methanogenic conditions are enhanced by biological pretreatment leading to an overall reduction in the time period of gas emissions. The discrepancy between these studies appears to depend on local factors and different landfilling techniques also presumably many are in the laboratory. In the field, difficulties of accessing the waste by the recirculating liquid remain a problem.

MBP plants have been in operation for only a few years. As a result, MBT waste studies are still at an early stage and further research is required. Available data suggest that pretreatment is a viable option for reducing the gas generating potential and leachate strength. However, no studies have captured the complete stabilisation process. Biogas and leachate characteristics are reported in separate studies and do not provide information on the releasing behaviour of heavy metals in leachate. Moreover, the mass balance approach was not used in any of these studies to confirm the reliability of data and examine the transformation of carbon and nitrogen from solid to liquid and gaseous phases.

2.5 Waste settlement in MSW landfills

Knowledge of settlement is vital for designing landfill final cover and post closure developments over closed landfills. Post closure settlement leads to

problems like crack formation in the cover and liner system, damage to landfill infrastructure e.g. gas and leachate extraction systems.

2.5.1 Settlement stages and mechanism

Waste settlement may be classified into three main stages commonly referred to as: immediate compression, primary settlement and secondary settlement (Powrie et al., 2009; Wall & Zeiss, 1995; Morris & Woods, 1990). Immediate compression is the instantaneous settlement that occurs after an external load is applied to the waste. It probably results from the expulsion of air initially present in the voids, together with the compression of air and/or certain types of particles or materials. Primary settlement arises from the consolidation of the waste as water flows out from the pores in response to the load. Secondary settlement occurs over a longer period of time and is generally associated with the waste biodegradation as well as mechanical creep. The settlement of a biodegradable waste in a landfill is a complex process and arises from a variety of mechanisms. Powrie et al. (2009) described the mechanisms controlling the three stages of settlement in more detail than earlier work by Hudson et al., (2004) and Edil et al., (1990):

1. Sliding, reorientation or distortion of waste particles as vertical stresses are increased
2. Compression of the pore fluid
3. Compression or crushing of the waste particles
4. Breakage of particles with increasing stress or softening of particle contacts on wetting resulting in a loss of strength and/or structure
5. Degradation due to biological decomposition and physico-chemical processes

6. Mechanical creep i.e. continuing settlement at constant effective stress

2.5.2 Settlement models

A reasonable prediction of settlement is important in estimating landfill capacity, designing final cover grades and gas and leachate extraction systems, and planning post closure redevelopment projects. Numerous models have been proposed to estimate settlement of the waste in landfills. A brief discussion of some of the more significant models is presented below.

2.5.2.1 Consolidation models

These models developed to estimate MSW settlement are based on the consolidation theory of soils (Sowers, 1973; Rao et al., 1977; Oweis & Khera, 1986). The Sowers (1973) model is a simplified one dimensional model that deals with primary and secondary settlement separately and requires separate equations to estimate waste settlement in both phases. This model assumes that the portion of settlement curve corresponding to the secondary settlement is linear with the logarithm of time as expressed by the equation:

$$\frac{\Delta h_s}{h_{ref}} = C_{\alpha\varepsilon} \log\left(\frac{t}{t_{ref}}\right) \qquad \text{Eq. 2.1}$$

$$\text{with } C_{\alpha\varepsilon} = \frac{C_\alpha}{(1+e_o)} \qquad \text{Eq. 2.2}$$

where, Δh_s= secondary settlement at time t, h_{ref} = height of waste upon completion of primary settlement, t_{ref} = a reference time for secondary settlement, $C_{\alpha\varepsilon}$ = coefficient of secondary settlement and e_o = initial void ratio

Based on field measurements, Sowers (1973) recommended values of $C_{\alpha\varepsilon}$ in the range 0.02 to 0.07. A value of 0.02 corresponds to unfavourable conditions to biodegradation, while 0.07 corresponds to favourable conditions for biodegradation.

The advantages with this model are that its formulation is simple and it involves only a few number of model parameters which can often be determined from simple compression tests. The disadvantage is that it is often difficult to make a distinction between primary and secondary settlement, while creep and biodegradation are lumped together. Another drawback with this model is that landfills are usually not saturated, therefore the classical consolidation model for saturated soils may not be appropriate.

Wall & Zeiss (1995) and El- Fadel & Al-Rashed (1998) used a one dimensional consolidation equation to model settlement in bioreactor landfill cells. Wall & Zeiss (1995) conducted a lab study and showed that secondary settlement is linear with the logarithm of time. They found one dimensional consolidation theory suitable to simulate the settlement. Their conclusion that decomposition marginally affects the rate of secondary settlement is probably incorrect as the experimental conditions were not conducive for biodegradation. El- Fadel & Al-Rashed (1998) demonstrated that a one dimensional consolidation model had a greater application potential for practice as it provided a better representation of laboratory and field settlement data compared to other models.

Olivier et al. (2003) performed laboratory one dimensional tests on MSW and proposed an Incremental Settlement Prediction Model based on the placement of the waste in elementary layers leading to the formation of total

height of the waste column. This algorithm integrates the behaviour in primary and secondary settlement of each elementary layer to determine the overall behaviour of the waste column, giving a better indication of long term settlement.

2.5.2.2 Empirical models

The rheological model (Gibson and Lo, 1961), power creep law model (Edil et al., 1990), logarithmic model (Yen and Scanlon, 1975) and hyperbolic model (Ling et al., 1998) were mainly designed to predict only the mechanical compression characteristics of soil like materials. Therefore they are not capable of estimating the settlement of MSW landfills, which results from both mechanical creep and biodegradation. These models are based on a single equation and do not require or allow separation of settlement into primary and secondary components. They attempt to simulate the settlement of a landfilled waste by a mathematical function involving adjustment of empirical parameters which are site specific and seldom have a physical significance.

Rheological and power creep law model are not based on biodegradation, so they do not realistically model settlement in a landfill. Studies conducted by Wall & Zeiss (1995) and El- Fadel & Al-Rashed (1998) showed that the one dimensional consolidation model was better than Gibson and Lo, and power creep law models. The Yen and Scanlon (1975) model cannot predict reliable values of settlements beyond a certain time and settlement values tend to be negative. Similarly, the Ling et al. (1998) model gives negative settlement for very large times indicating that the landfill undergoes expansion which is most unlikely if not impossible.

A major disadvantage common to most models is that they involve too many parameters, some of which are practically difficult to determine and may have no physical significance.

2.5.2.3 Biomechanical models

Edgers et al. (1992) developed a waste settlement model which takes into consideration the mechanical compression and decomposition through the microbiological processes within the landfill. They expressed biodegradation induced settlement as a function of the activity of microorganisms. The major drawback is that the model incorporates the growth kinetics of a single species of bacterial population (methanogens), and therefore, underestimates the role of hydrolysis which is the rate limiting step in degradation. Another limitation is that there is no accurate method available to predict the critical time at which the settlement rate starts to increase due to biological activity.

Park and Lee (1997) defined the concept of settlement that occurs due to the decomposition of biodegradable refuse using first order kinetics as follows:

$$\varepsilon_b = \varepsilon_{bt}\left(1 - e^{-k_b t}\right) \qquad \text{Eq. 2.3}$$

where, ε_b is the settlement strain due to biodegradation at a time t since the start of degradation, ε_{bt} is the total amount of settlement strain resulting from biodegradation and k_b is a first order degradation rate constant. This model assumes that the settlement process of biodegradable solid waste is due to the solubilisation from the decomposition. This model does not on its own account for mechanical compression during or immediately after waste placement, although it handles the biodegradation induced settlement well. Elagroudy et al. (2008) confirmed the applicability of Equation 2.3 to wastes

of different composition, with or without the addition of sewage sludge, in different operational conditions.

At present, due to the simplicity and familiarity of a consolidation based approach to practicing engineers, Sowers method is the most widely used for long term settlement prediction. Models incorporating separately mechanical creep and biodegradation are generally not available.

The Landfill degradation and transport (LDAT) model developed by the University of Southampton, UK incorporates microbial kinetics of leachate and gas production and their transport in addition to the consolidation of MSW (White et al., 2004; White et al., 2003). This model integrates several sub models (degradation, settlement, change in waste structure, leachate and gas generation and flow) to simulate all the spatially distributed processes occurring in a landfill. It uses the chemical pathways for the degradation of solid phase into liquid and then to a gas phase. The waste settlement is linked to biodegradation through loss of mass. The complexity of the model mirrors the biochemical processes involved in the degradation of organics in solid waste.

The advanced model, HBM (Hydro-Bio-Mechanical model, Mc Dougall, (2007)) is a coupled framework for the integrated analysis of the hydraulic, biodegradation and mechanical behaviour of landfilled MSW. The biodegradation model is a two stage anaerobic digestion model in which VFA and methanogenic biomass concentrations interact to control mineralisation of organic matter. The mechanical model combines load, creep and biodegradation induced effects to predict landfill settlement. It is the distinction between time dependent creep and rate limited

biodegradation effects that differentiate LDAT and HBM models from earlier settlement models. These models are complex in their formulation and remain relatively complex tool to implement.

Machado et al. (2002) proposed an advanced constitutive model to simulate the mechanical behaviour of MSW based on laboratory experiments. They assumed that the behaviour is controlled by two distinct parts: fibrous material and organic paste, according to a coupled elasto-plastic model. In a follow up study, Machado et al. (2008) combined a mechanical creep component together with biodegradation induced settlement related to gas generation. The effect of biodegradation and associated mass loss is included in the model through a first order decay model.

Despite the fact that these models take into consideration most of the mechanisms, the requirement for a large number of parameters makes them at present difficult to implement in practice.

Recently, Gourc et al. (2010) developed a one dimensional biomechanical model to predict secondary settlement due to mechanical creep and biodegradation mechanisms. This model has separate equations to distinguish mechanical and biodegradation settlements. Biodegradation induced settlements were evaluated from the biogas production potential assuming the amount of settlement is directly proportional to the amount of solids solubilised.

2.6 Previous studies on waste settlement

Watts and Charles (1990, 1993) carried out field studies of the settlement of recently placed domestic refuse at two large disposal sites in the south of

England. At the Brogborough site the waste compressed by 11% over a period of 42 months. At Calvert, the corresponding compression was 4% only 11 months after landfill completion. On the basis of their results, Watts and Charles (1990, 1993) concluded that recently placed domestic refuse landfill is subject to large reductions in volume as a result of biodegradation, and that it is highly compressible under applied load. The authors observed that it is possible to distinguish the creep component of settlement from the biodegradation component.

There are a few studies of settlement in bioreactor landfills in the general literature. One of the first studies was presented by Wall and Zeiss (1995), where six laboratory scale reactors were used to model both settlement and decomposition over a period of 225 days. Three cells were designed to simulate bioreactor landfills (enhanced cells), while the other three were simulated dry landfills. The initial compression on load application was 17% and 26% in the dry and enhanced cells respectively. The primary settlement resulted in a further settlement of 12% and 15% with secondary settlements of 2% and 4% in the dry and enhanced cells respectively. El Fadel and Al Rashed (1998) analysed five years of settlement data from the Mountain View Landfill, California. The test cells operated with leachate recirculation settled 13 to 15%, while control dry cells settled 8 to 11%. Yuen and Styles (2000) presented the findings of a settlement investigation at an MSW landfill in Melbourne, Australia. One of the objectives of the project was to evaluate the full scale landfill settlements on two different sections of the landfill, one operated as a bioreactor landfill with leachate recirculation, and the other operated as a dry landfill. After approximately 3.5 years of monitoring, the settlement in the bioreactor landfill section reached 6%, while for the dry

landfill the settlement reached about 3%. For the Yolo County test cells, Mehta et al. (2002) reported 15.5% settlement in the enhanced cells as against 3% in the control cells over a period of just less than 3 years.

Benson et al. (2007) analysed five full scale landfills in North America operating as bioreactors. It was found that the waste in landfills settled by 22-25% over a period of 1000 days. They indicated that settlements are larger and occur much faster in landfills operated as bioreactors. Settlement as a fuction of compression and biodegradation was studied by Olivier and Gourc (2007). They conducted large scale laboratory experiments on MSW subjected to a vertical pressure of 130 kPa over a period of about 2 years. The primary and secondary settlements were reported as 25.4% and 23.9% respectively. Bareither et al. (2008) studied the influence of biological activity on compression of MSW in lab scale reactors and reported a total settlement of 34% in biological test cells as against 31% in a non biological cell.

The behaviour of secondary settlement is complicated due to the combined effects of mechanical creep and biodegradation. Several studies have evaluated a coefficient of secondary settlement ($C_{\alpha\varepsilon}$) describing continuing settlement with time under constant effective stress. Most researchers do not make a clear distinction between creep and biodegradation, and evaluated $C_{\alpha\varepsilon}$ based on the similarity of the time dependent nature of creep and biodegradation induced settlements. $C_{\alpha\varepsilon}$ was determined from the slope of the settlement versus log time plot assuming that biodegradation continued with time just like mechanical creep. This assumption is invalid as biodegradation cannot continue indefinitely with time. Therefore,

biodegradation cannot be lumped together with creep and the contribution of each mechanism must be isolated.

The values of $C_{\alpha\varepsilon}$ reported in the literature mostly include components of both creep and biodegradation. Edgers et al. (1992) reported that creep coefficients varied from 0.01 to 0.04 (average 0.025) and combined creep and biodegradation coefficients varied from 0.02 to 0.5 (average 0.1). Using data from the Mountain View bioreactor landfill, El-Fadel and Al Rashed (1998) reported creep coefficients of 0.015-0.035 and biological coefficients of 0.1-0.32 which included creep. Hossain et al., (2003) evaluated the secondary compression indices representing creep and biodegradation in the range 0.02-0.03 and 0.02-0.19 respectively. Bareither et al. (2008) reported secondary compression coefficients (combined creep and biodegradation) in the range 0.01-0.19. A brief summary of the range of published values of $C_{\alpha\varepsilon\ for}$ MSW is presented in Table 2.5.

A review of published data shows typical values of $C_{\alpha\varepsilon}$ in the range 0.01 to 0.32. The wide variation in $C_{\alpha\varepsilon}$ values is attributed to various factors e.g. the extent of degradation, moisture content, density, test duration and equipment.

Table 2.5: Summary of published values of $C_{\alpha\varepsilon}$ for MSW

Source	$C_{\alpha\varepsilon}$
Sowers (1973)	0.02 to 0.07
Rao et al. (1977)	0.012 to 0.046
Oweis and Khera (1990)	0.02 to 0.072
Wall and Zeiss (1995)	0.033 to 0.056
Gabr and Valero (1995)	0.015 to 0.023
Green and Jamenjad (1997)	0.01 to 0.08
El Fadel and Al Rashed (1998)	0.015 to 0.32
Landva et al. (2000)	0.01 to 0.016
Machado et al. (2002)	0.012 to 0.016
Hossain et al. (2003)	0.02 to 0.19
Durmusoglu et al. (2006)	0.043 to 0.083
Bareither et al. (2008)	0.01 to 0.19

Current knowledge regarding settlement characteristics is mainly related to MSW landfills. There are virtually no settlement data for MBT waste landfills. The behaviour of MBT waste will be different from the experiences which have been gained in the past for MSW landfills, and a clear distinction between the creep and biodegradation components of settlement is needed. The current study will distinguish creep and biodegradation settlement in MBT waste and analyse them separately based on simplified models.

2.7 Hydraulic properties of MSW in landfills

Knowledge of the hydraulic behaviour of waste will assist in the accurate determination of leachate collection and distribution rates and volumes. Hydraulic conductivity is a key parameter for the management of leachate. Hydraulic conductivity in MSW depends mainly on the pore size and geometry, which in turn varies with the size and shape of individual particles and packing density. The drainable porosity determines the void space available for water movement through the saturated waste. Previous research (e.g. Powrie and Beaven, 1999; Reddy et al., 2009; Durmusoglu et al. 2006; Landva and Clark, 1990) has demonstrated that the hydraulic conductivity of MSW is controlled by the vertical stress, through its impact on waste compression and density.

Powrie and Beaven (1999) carried out tests on household waste in a large scale compression and reported a decrease in hydraulic conductivity as a function of applied stress and density. This trend of decrease in hydraulic conductivity with increasing density and/or applied stress was later corroborated by work performed by Reddy et al. (2009), Olivier & Gourc (2007), Durmusoglu et al. (2006) and Stoltz et al. (2010).

Reddy et al. (2009) demonstrated that the hydraulic conductivity of MSW can be significantly influenced by vertical stress which was attributed to the increase in density leading to low void ratio. Table 2.6 summarises the range of hydraulic conductivity values reported in previous studies on MSW. Other factors influencing the hydraulic conductivity are the gas entrapped in the waste (Hudson et al., 2004; Jain et al., 2006) and the presence of plastic sheet fragments in the waste (Xie et al., 2006).

Table 2.6: Summary of the reported values of hydraulic conductivity of MSW

Reference	Hydraulic conductivity (m/s)	Dry density (kg/m^3)	Test
Laboratory tests			
Bleiker et al. (1995)	$1x10^{-6}$ - $5x10^{-9}$	500 – 1200	Falling head
Chen & Chynoweth (1995)	$9.6x10^{-4}$ - $4.7x10^{-7}$	160 – 480	Constant head
Powrie & Beaven (1999)	$1.5x10^{-4}$ - $3.7x10^{-8}$	390 – 720	Constant head
Durmusoglu et al. (2006)	$1.2x10^{-4}$ - $4.7x10^{-6}$	N/A	Falling head
Olivier & Gourc (2007)	$1x10^{-4}$ - $1x10^{-6}$	490 – 710	Falling head
Reddy et al. (2009)	$2x10^{-3}$ - $7.8x10^{-7}$	320 – 960	Constant head
Staub et al. (2009)	$7.4x10^{-5}$ - $4.6x10^{-6}$	370 – 530	Falling head
Stoltz et al. (2010)	$1x10^{-4}$ - $1.1x10^{-5}$	600 – 900	Constant & falling head
Field tests			
Oweis et al. (1990)	$2.4x10^{-5}$ - $9.4x10^{-6}$	680	Pumping test
Landva & Clark (1990)	$4x10^{-4}$ - $1x10^{-5}$	1000 – 1400	Flow nets
Jain et al. (2006)	$6.1x10^{-7}$ - $5.4x10^{-8}$	N/A	Borehole permeameter
Machado et al. (2010)	$1x10^{-5}$ - $1x10^{-8}$	N/A	Borehole infiltration

The hydraulic conductivity may also depend on the extent of degradation of waste causing a possible change in the composition and size of the waste particles. However, very few data are available on variation of hydraulic conductivity with degradation. Powrie & Beaven (1999) analysed hydraulic conductivity of fresh, processed, and aged waste and showed that differences in the values of hydraulic conductivity resulting from particle size reduction and waste degradation are less significant than the effects of stress and compression.

Hossain et al. (2009) investigated the change in hydraulic conductivity of MSW at different stages of biodegradation in a laboratory bioreactor. They reported a decrease in hydraulic conductivity from 8.8 $x10^{-5}$ m/s to 1.3$x10^{-5}$ m/s which was attributed to a decrease in particle size.

Trends of decreasing porosity with increasing density and/or applied stress are highlighted in studies by Powrie & Beaven (1999), Hudson et al. (2004), Staub et al. (2009), and Stoltz & Gourc (2007). These studies have reported total and effective porosities in the range 45-65% and 1.5-25% respectively. Stoltz & Gourc (2007) studied shredded MSW and reported total porosity values of 45% and 62% at dry densities of 770 and 540 kg/m^3 respectively.

Several authors have studied the decrease in hydraulic conductivity of MBT waste as a function of density, as summarised in Table 2.7. Kuehle-Weidemeier (2004) reported a range of hydraulic conductivity from 3$x10^{-6}$ at 50 kPa to 6$x10^{-9}$ m/s at 550 kPa. These values are nearly of the same order of magnitude to those derived for MSW and as such indicate a comparable hydraulic conductivity.

Table 2.7: Previous studies reporting hydraulic conductivity of MBT waste

Reference	Hydraulic conductivity (m/s)	Density (kg/m^3)
Bidlingmaier et al. (1999)	$1x10^{-4}$ - $1x10^{-7}$	600 – 900 (dry)
Bauer et al. (2006)	$1x10^{-5}$ - $1x10^{-7}$	850 – 1200
Kuehle-Weidemeier (2004)	$3x10^{-6}$ - $6x10^{-9}$	690 – 920 (dry)
Xie et al. (2001)	$2x10^{-7}$ - $6.3x10^{-8}$	800 – 1100 (dry)

CHAPTER 3 MATERIALS AND METHODS

3.1 Waste samples studied

Two different waste materials were studied: UK MBT and German MBT. They originate from two different MBP plants, one in the UK and one in Germany.

3.1.1 MBT waste from Southern England (UK MBT waste)

A sample of about 500 kg of MBT waste was obtained from White's Pit waste processing plant, a mechanical-biological treatment facility in Southern England. This facility includes a shredder, conveyor belts, magnets, screens and windrows for aerobic composting. In the mechanical stage, waste was sorted to extract recyclable material. The remaining waste was then broken down into smaller parts by shredding and screened followed by recovery of ferrous metals. Thereafter, the waste was aerobically composted in forced aerated windrows with regular wetting and turning in fully enclosed halls for a period of six weeks. After processing, the material was screened again to extract any remaining dry recyclables, giving a maximum particle size for the residual waste of about 20 mm. The flow diagram of MBP process is shown in Figure 3.1.

3.1.2 MBT waste from Northern Germany (German MBT waste)

A sample of about 120 kg of MBT waste was obtained from Hannover Waste Treatment Centre, a mechanical-biological treatment facility in Hannover,

Northern Germany. During the mechanical stage of the process, waste was sorted, shredded and screened, and recyclable materials and metals were removed. The high calorific value fraction (mainly non-recycled paper and plastics) was sent as a refuse derived fuel to the incineration plant. Waste was then anaerobically digested in fermentation tanks for a period of 3 weeks. The digested material was extracted and transferred to the aerobic post treatment area where it was composted in enclosed windrows for about 6 weeks. The maximum particle size of the residual material was about 60 mm. Photographs of the unsorted waste are shown in Figures 3.2 and 3.3 for the UK and the German MBT wastes respectively.

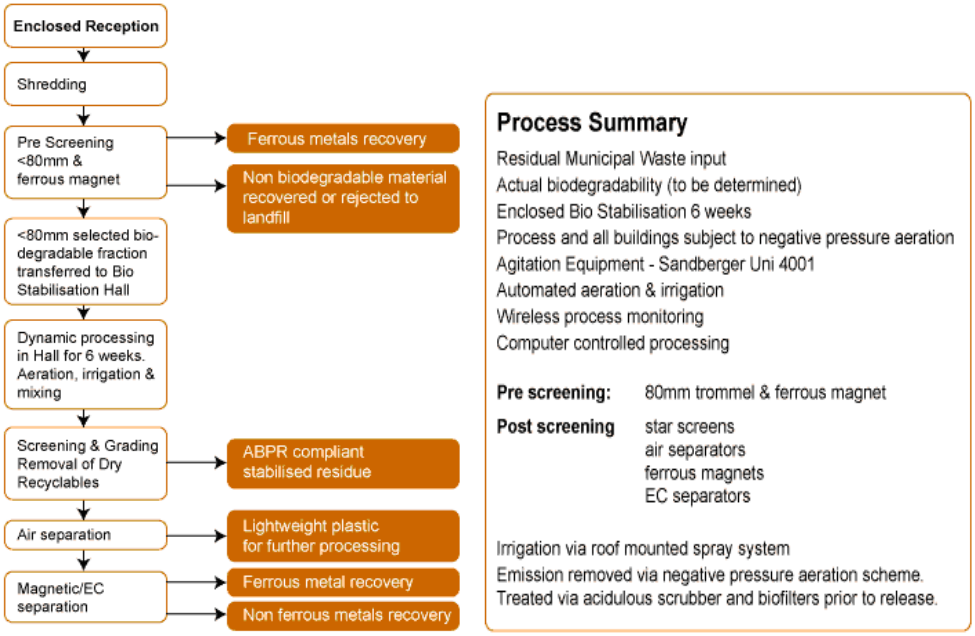


Figure 3.1: MBP process flow diagram (after New Earth Solutions: http://www.newearthsolutions.co.uk/residual-waste-treatment/process-description/)

3.2 Waste characterisation

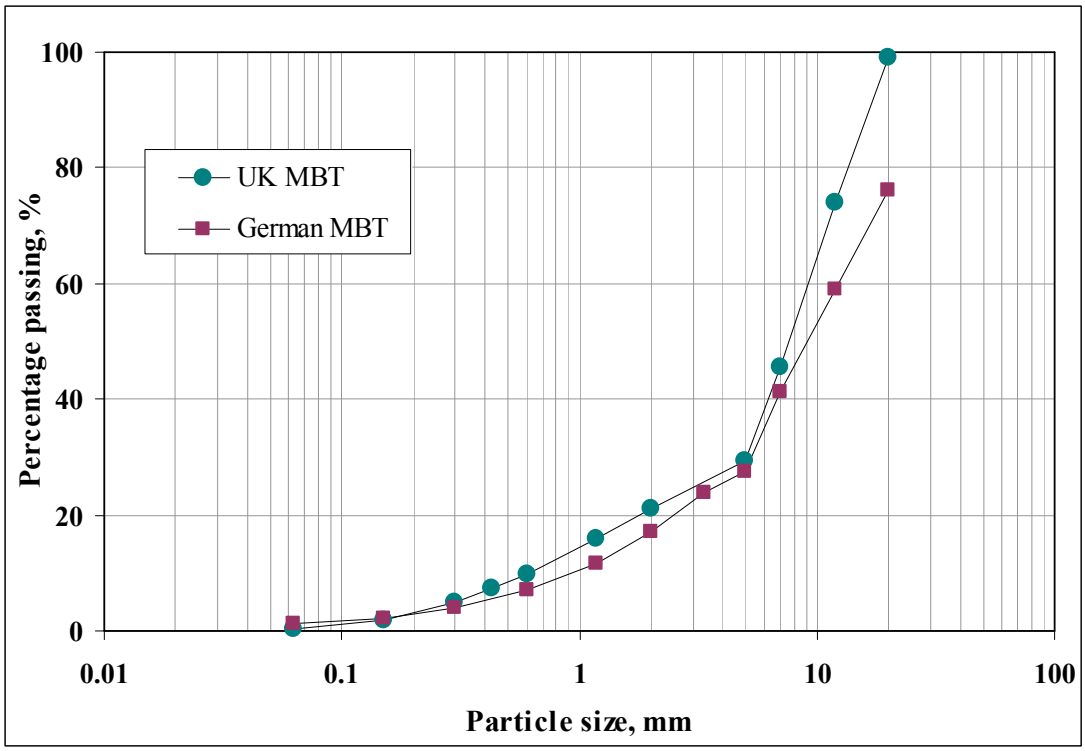
A detailed physical and chemical characterisation of the MBT waste was undertaken as a part of a wider study into its mechanical, hydrogeological and biological behaviour.

3.2.1 Particle size distribution

Representative samples of about 25 kg of both the UK and the German MBT wastes were prepared by quartering the bulk sample. These samples were then sieved mechanically through a set of sieves of sizes 20, 12, 7 and 5 mm into five fractions having particle size ranges: >20 mm, 20-12 mm, 12-7 mm, 7-5 mm and <5 mm. Both wastes were sieved at their as-received water contents, but the German MBT had been oven dried for transport. Further analyses of particle size distribution were carried out in accordance with BS 1377-2 (1990), using about 1 kg of the dried material fraction <5mm from both MBT wastes. The particle size distribution curves for both MBT samples are given in Figure 3.4. There were no significant differences between the results for the two wastes except that the German MBT waste had a higher fraction of large size particles and slightly less fine material than the UK MBT waste.

Figure 3.2: MBT waste sample as received from White's Pit waste processing plant (UK MBT waste)

Figure 3.3: MBT waste sample received from Hannover waste treatment centre (German MBT waste)



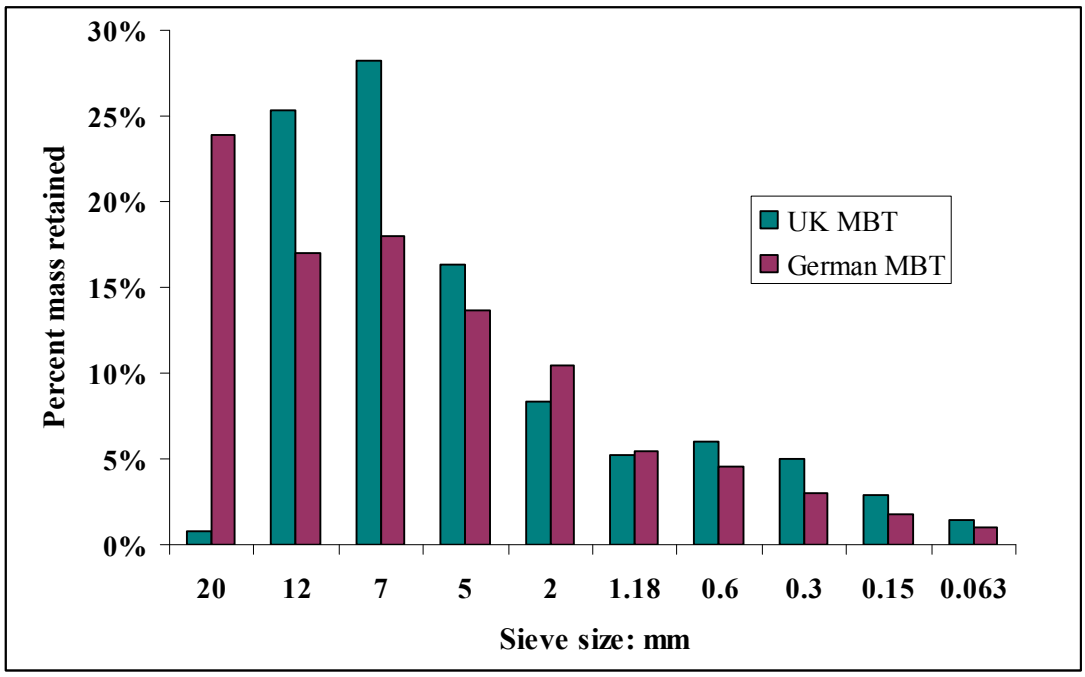


Figure 3.4: Particle size distribution of the UK and German MBT wastes

3.2.2 Waste composition

Each size fraction (except that less than 5 mm) from each MBT waste was sorted manually into various material categories i.e. flexible plastics, rigid plastics, textiles, paper, glass, wood, bones, rubber, ceramics, metals, stones and unidentified >5 mm. All material <5 mm was placed in the "unidentified <5mm" category. The "unidentified" category represents a mixture of different components that could not be identified or further separated. Material less than 5 mm could not be sorted further due to its small particle size. The unidentified >5 mm category consists of particles greater than 5 mm whose material could not be identified, usually because they were encased in soil-like material which was impossible to remove without breaking the particle.

The composition of each waste sample by material type (expressed as percentage of the total dry weight) is given in Table 3.1. The majority of the waste samples was visually unidentifiable. This is evident from the material placed in unidentified category making up more than 50% of the waste. Glass (22.8% in the UK and 24.3% in the German MBT waste) was the second most prevalent material. Rigid and flexible plastics together accounted for about 10% of each MBT sample. A slight difference between the UK and German MBT wastes could be a result of the different processing of waste in the pretreatment stages.

The composition of raw MSW used in an earlier study on biodegradation and settlement (Ivanova et al., 2008*a*, 2008*b*) is given in Table 3.2.

Table 3.1: MBT waste components expressed as dry weight percentages

Component	Percentage of dry mass (%)	
	UK MBT waste	German MBT waste
Paper	0.43	0.18
Flexible Plastics	4.57	2.4
Rigid Plastics	6.27	5.91
Wood	1.57	3.22
Textile	1.33	0.63
Rubber	0.18	0.25
Bones	0.27	0.37
Metal	0.49	1.49
Ceramics	2.29	4.25
Stones	1.73	3.17
Glass	22.77	24.36
Unidentified >5 mm	28.95	26.75
Unidentified <5 mm	29.15	27.02
Total	100.00	100.00

Table 3.2: Raw MSW waste composition (Ivanova et al. 2008*a*, 2008*b*)

Component	Dry weight, kg	% by dry weight
Paper	52.0	27.3
Light plastic	19.3	10.1
Heavy plastic	18.8	9.9
Wood	6.1	3.2
Yard waste	35.0	18.4
Food	4.3	2.3
Textile	5.9	3.1
Combustible[1]	5.6	3.0
Metal	12.9	6.8
Glass	5.0	2.6
Others<10mm	25.3	13.3
Total	190.2	100.00

Note:[1]Combustible fraction of the waste includes leather, rubber, wipes, and disposable nappies.

3.2.3 Moisture content

The moisture content of the MBT waste was determined by oven-drying the representative samples at 70°C to a constant weight. The moisture content is defined as the ratio of mass of water to the mass of waste solids and can be reported in two ways: the wet-weight basis or the dry weight basis given by

$$w_{dry} = \frac{(a-b)}{b} \quad \text{and} \qquad \text{Eq. 3.1}$$

$$w_{wet} = \frac{(a-b)}{a} \qquad \text{Eq. 3.2}$$

where, a= initial mass of sample, and

b= dried mass of sample

The average moisture content of the UK MBT was found to be 36.1% of the dry weight. The German MBT sample was received oven-dried, so its moisture content was not determined.

3.2.4 Density and porosity

The dry density, bulk density, porosity and drainable porosity were determined using the following definitions:

Dry density: The initial dry density of the waste was determined as the mass of dry solids divided by the total volume of waste in the reactor described in Section 3.3.1. Dry density (ρ_d) is sometimes used as an indication of the degree of compaction of the waste.

Bulk density: The bulk density (ρ) was calculated as the ratio of the total mass of waste (including moisture) to the volume of the waste in the reactor. The dry density is related to bulk density as:

$$\rho_d = \rho/(1+w_{dry}) \quad \text{Eq. 3.3}$$

Total porosity: Total porosity (n) is defined as the volume of voids per unit total volume given by

$$n=V_v/(V_s+V_v) \quad \text{Eq. 3.4}$$

where V_v= volume of the voids and V_s= volume of solids

Drainable porosity: This is a measure of the volume of water that will drain from a fully saturated material under the influence of gravity. The drainable porosity (n_d) is defined as the volume of drainable voids (V_d) per unit total volume

$$n_d = V_d/(V_s+V_v) \quad \text{Eq. 3.5}$$

The detail on measurement of drainable porosity is described in Section 3.5.1.1. The parameters ρ, ρ_d, n and n_d were determined in experiments on both the UK and the German MBT wastes at 0, 50 and 150 kPa loading and the results are given in Section 5.3.1.

3.2.5 Loss on ignition

Representative samples (100 g) of each MBT waste were prepared containing the same proportions of each component as the original sample (Table 3.1). Prior to the analyses for loss on ignition, total carbon and total nitrogen, all

non grindables (metal, glass, ceramic and stone) were removed from the sample. The remaining waste was dried at 70°C and milled to a fine powder using a Foss Knifetec 1095 mill in conjunction with a Foss Cyclotec 1093 mill. All solids preparation and analysis were performed in triplicate and average values are reported in Table 3.3.

Loss on ignition (LOI), also known as volatile solids content, generally represents the total organic content of the waste. LOI content was measured as the weight loss from the dried sample on ignition at 550°C in a muffle furnace (Carbolite, UK) for two hours, divided by the initial dried sample weight.

3.2.6 Total carbon and total nitrogen

The total carbon and total nitrogen contents of the samples were measured using a CE Instruments Flash EA 1112 Elemental Analyser (Thermo Finnigan, Italy). The equipment works on flash combustion of the sample in a gas flow temporarily enriched with oxygen at a temperature greater than 1700°C. The released oxides of nitrogen and carbon (depending on the composition of the sample) in the gas mixture was then analysed by gas chromatography, with the different components being measured by an appropriate detector.

3.2.7 Fibre Analysis

Representative samples (100g) of each MBT waste were prepared in the way described in Section 3.2.5. Prior to the analysis, all plastics and non grindable components were removed and the remaining waste was then dried and

milled as before. Different fibre fractions: cellulose, hemicellulose and lignin were analysed in triplicate and the average values are reported in Table 3.3.

Fibre analysis was performed using the Foss technology system FibreCap technique, which is an improved procedure based on standard methods outlined by Van Soest (1967). The apparatus used was Foss Analytical FibreCap 2021/2023 system (Kitcherside et al., 2000), as shown in Figure 3.5. The test includes three separate procedures of acid detergent fibre (ADF), neutral detergent fibre (NDF) and acid digestible lignin (ADL) analysis to segregate cellulose (C), hemicellulose (H) and Lignin (L) in the test samples. The analyses were originally formulated for forage and animal feeds (Van Soest et al., 1991). Each analysis includes procedures of digestion in heated solutions, drying the residue and determining the ash content.

Figure 3.5: Foss Analytical FibreCap system for fibre analysis

The sample was weighed into a FibreCap capsule and an NDF test was carried out using a neutral solution made from sodium lauryl sulphate, EDTA, sodium tetra Borate dicahydrate and disodium hydrogen phosphate that removes starch, protein, organic acids etc. α-Amylase was also used to improve the solubilisation of starch. The fibre residue obtained, NDF, is composed of cellulose, hemicellulose and lignin. The ADF test requires digestion of the sample using 0.5 M sulphuric acid containing 2% CTAB (Cetyltrimethyl ammonium bromide), for 1 hour after the solution reaches boiling point. The fibre residue obtained, ADF, composed of only cellulose and lignin. The ADL analysis requires that the sample first subjected to the ADF test and the residue left is further treated with 72% (w/w) sulphuric acid for a period of 4 hours to dissolve the cellulose, leaving only lignin as residue. After digestion in the various solutions, in each analysis, the residues remaining in the capsule were dried at 105°C for at least 5 hours in an oven. These dried residues and capsules were combusted at 600 °C for 4 hours in a muffle furnace. The dry samples and ash after ignition were weighed. The components determined from these tests is summarised below:

NDF = Cellulose + Hemicellulose + Lignin + Mineral ash Eq. 3.6

ADF = Cellulose + Lignin + Mineral ash Eq. 3.7

Cellulose = ADF – ADL Eq. 3.8

Hemicellulose = NDF – ADF Eq. 3.9

ADL = Lignin + Mineral Ash Eq. 3.10

These tests were used successfully by Zheng et al. (2007) and Ivanova et al. (2008*b*) in characterizing the waste samples and assessing the waste biodegradability.

Table 3.3: Elemental and fibre analysis data for the UK and German MBT wastes

Chemical analysis	Concentration (% dry mass)	
	UK MBT waste	German MBT waste
Cellulose, %	10.24	7.96
Hemicellulose, %	4.54	3.91
Lignin, %	12.63	13.01
(C+H)/L ratio	1.17	0.91
Total carbon, %	22.68	19.85
Total nitrogen, %	1.81	1.52
Loss on ignition, %	42.91	34.84

3.3 Equipment

Large scale consolidating anaerobic reactors and small scale biochemical methane potential reactors were set up in the laboratory to study the biodegradation and settlement behaviour of MBT waste.

3.3.1 Large scale Consolidating Anaerobic Reactor

The consolidating anaerobic reactor (CAR) comprises a Perspex cylinder 480 mm diameter, 900 mm tall, and a load delivery system that can apply a constant vertical stress to the waste. The load is applied to the top of the

waste using a hydraulically operated ram attached to a 475 mm diameter perforated metal platen maintained in contact with the waste sample. This platen moves with the waste as it settles as a result of loading, degradation and creep, maintaining a constant load throughout the testing period. The load platen was perforated to allow leachate to pass through the waste column. The CAR is designed to simulate anaerobic conditions in a landfill, and a maximum vertical load of 150 kPa (equivalent to 15 m depth of landfill, assuming the average unit weight of waste in landfills as 10 kN/m^3 (Beaven, 2000)) can be applied to the waste. A schematic view of a CAR is shown in Figure 3.6.

A 10 cm gravel drainage layer (13.5 to 20 mm particle size) was placed at the base of the reactor (CAR) followed by a geotextile membrane (polypropylene square mesh 2mm x 2mm). The gravel used was washed with distilled water and dried prior to placement in the CAR. The purpose of the geotextile was to separate the waste from the gravel, avoid the migration of solid particles with the leachate and prevent clogging of the drainage system. The MBT waste (which had been dried in an oven at 70°C) was placed in the CAR in 10 successive layers of 4 kg each compacted to a thickness of 5 cm by hand tamping with a 350 mm diameter wooden tamper to ensure a uniform density throughout the reactor. This was checked by measuring the thickness of each layer of known mass, after placement and tamping. A further layer of gravel (5 cm thick) was placed to give an even distribution of leachate over the top surface of the waste, and was again separated from the waste by a geotextile membrane. The CAR was placed within a compression loading rig, and ports located on the side of the reactor were used as piezometers and for recirculating leachate from the bottom to the top.

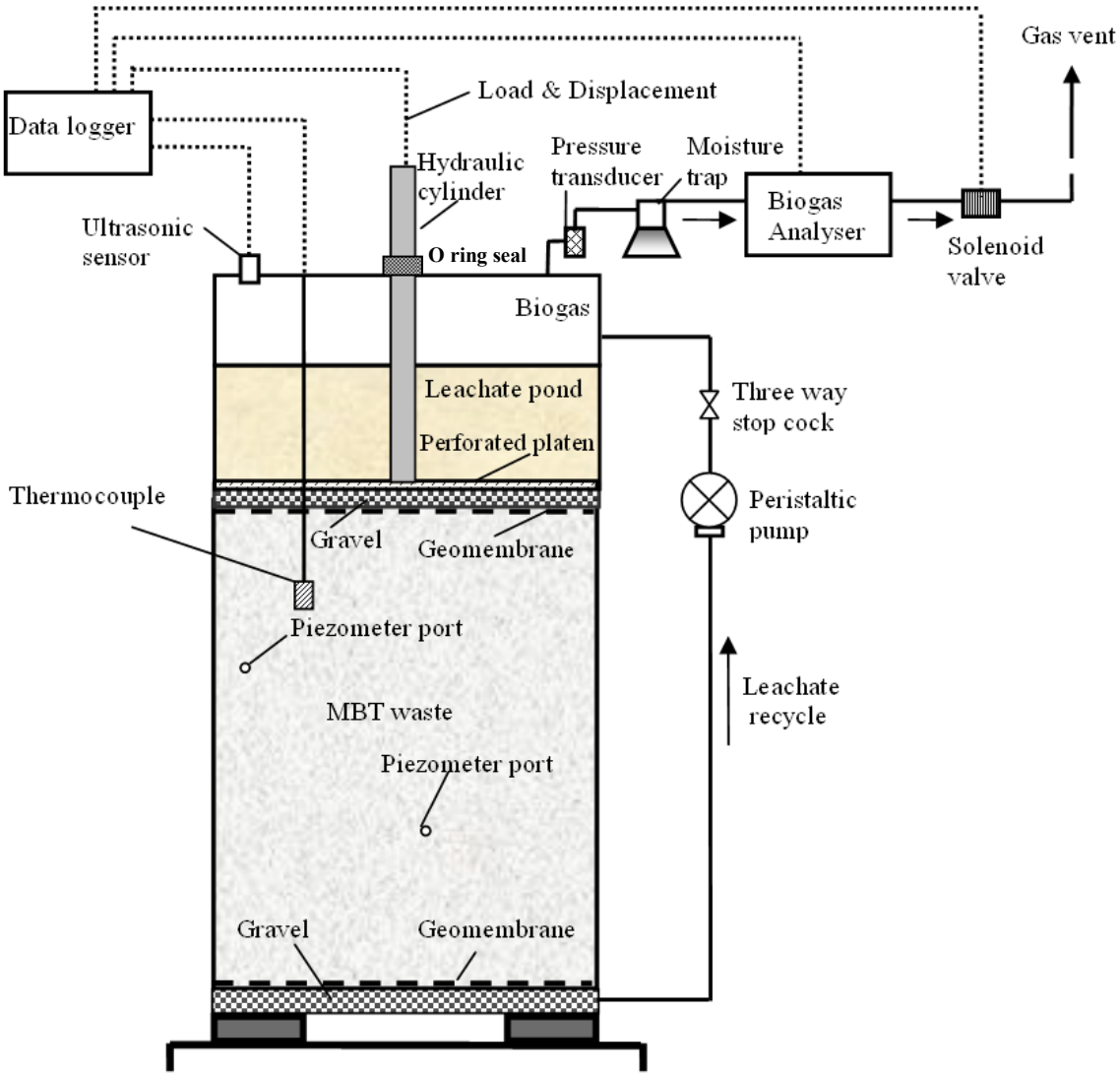


Figure 3.6: Schematic view of the consolidating anaerobic reactor (CAR)

Long term testing of each MBT waste was undertaken using two CARs: a control reactor (CAR1) and a test reactor (CAR2). These CARs were equipped with thermocouples to monitor the temperature of the waste, a linear variable displacement transducer (LVDT) to monitor waste settlement,

pressure transducers to record load, a gas measurement system and a leachate recirculation system comprising a peristaltic pump. The load, settlement, temperature and biogas outflow were monitored continuously by an automatic logging system. Ultrasonic sensors installed to monitor the leachate level in the CARs failed to work.

Biogas production was determined by allowing the gas to build up in the headspace volume (V_h) to a small positive pressure (Δp) above ambient atmospheric pressure (p_a) as measured by a pressure sensor. The biogas within the reactor was vented automatically to atmosphere via a solenoid valve triggered by the data logging system in response to the limiting pressure rise Δp measured by the pressure sensor. The data logging system recorded each time the valve was activated. The volume of gas (V_g) released at each venting event was then calculated using the ideal gas equations.

$$p_1 V_1 = p_2 V_2 \quad \text{at constant temperature} \qquad \text{Eq. 3.11}$$

$$(p_a + \Delta p) V_h = p_a (V_h + V_g) \qquad \text{Eq. 3.12}$$

$$V_g = (\Delta p / p_a) \times V_h \qquad \text{Eq. 3.13}$$

The biogas volume determined in this way was verified by means of gas sampling bag and a water displacement technique.

The measured volume of biogas produced was corrected to dry gas at standard temperature and pressure (STP) using ambient temperature and pressure measurements and values of water vapour pressure. The term vapour pressure is used here to mean the partial pressure of water vapour in the atmosphere and was calculated from the ambient temperature using the equation (Buck, 1981).

$$p_w = 0.61121 \exp[17.502T/(240.97+T)] \quad \text{Eq. 3.14}$$

where T is the reference temperature in the laboratory in °C.

All biogas readings were standardised to dry gas at STP using

$$V'_g = V_g \frac{p_a}{p'_a} \times \frac{K}{K+T} \times \left(1 - \frac{p_w}{p_a}\right) \quad \text{Eq. 3.15}$$

where, p'_a is atmospheric pressure at sea level in kPa (101.3 kPa), and K is the absolute zero temperature in Kelvin (273.2 K).

3.3.2 Small scale Biochemical Methane Potential Reactors

BMP reactors were made from 1000 mL Nalgene bottles attached to a gas collection system that allowed the volume of gas produced during anaerobic degradation to be measured. The test included 12 BMP reactors (B1 to B12) each filled with 140 g representative sample of dried MBT waste containing the same proportions of the component composition as the original sample (Table 3.1). The waste samples were shredded to a maximum particle size of 10 mm prior to placement in the reactor vessels. Biogas production was measured using the displacement method by collecting the gas produced within an inverted glass burette containing water that had been acidified to prevent dissolution of biogas (Environment Agency 2005, Godley et al., 2007). A schematic view of the BMP apparatus and the experimental set up for the BMP reactors used in this study is shown in Figure 3.7. The measured volume of biogas produced was corrected to dry gas at STP using ambient temperature and pressure measurements and values of water vapour pressure as described earlier.

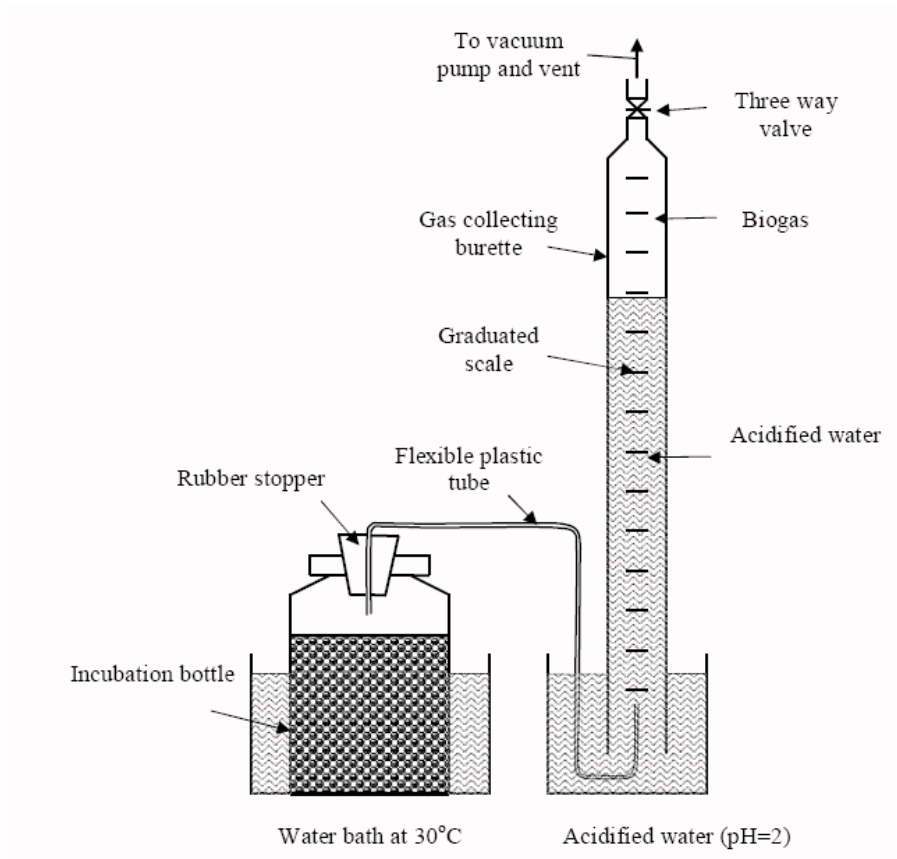


Figure 3.7: A schematic view and set up of biochemical methane potential test reactors

3.4 Synthetic leachate

To enhance the decomposition of the waste by anaerobic bacteria, synthetic leachate containing mineral nutrients and trace elements dissolved in deionised water as described by Florencio et al. (1995) was used in this study and details are given in Table 3.4.

Table 3.4: Recipe of synthetic leachate

Reagent	Concentration (mg/litre)	Reagent	Concentration (mg/litre)
$K_2HPO_4.3H_2O$	330.000	$MnCl_2.4H_2O$	0.500
NH_4Cl	280.000	$CuCl_2.2H_2O$	0.038
$MgSO_4.7H_2O$	100.000	$(NH_4)6MoO_{24}.4H_2O$	0.050
$CaCl_2.2H_2O$	10.000	$AlCl_3.6H_2O$	0.090
$FeCl_2.4H_2O$	2.000	$NiCl_2.6H_2O$	0.142
H_3BO_3	0.050	$Na_2SeO_3.5H_2O$	0.164
$ZnCl_2$	0.050	$CoCl_2.6H_2O$	2.000
EDTA	1.000		

3.5 Experimental procedures

3.5.1 Consolidating anaerobic reactors

In addition to the biodegradation and settlement behaviour, the hydraulic properties (hydraulic conductivity and drainable porosity) of each MBT waste were characterised, for which the testing methodology was as follows:

- Hydraulic properties i.e. drainable porosity (n_d) and saturated hydraulic conductivity (k) at the initial stage were determined in the CARs under no applied load (zero vertical stress). Other parameters such as n, ρ_d and ρ were also determined at this stage.

- A load of 50 kPa was applied in both the CARs and the parameters n_d, k, n, ρ_d and ρ were determined after 24 hours*.

- The CARs were completely sealed and appropriate valves were installed for gas and leachate handling. CAR2 was filled with leachate containing sludge seed and CAR1 with leachate comprising acids as detailed in Section 3.5.1.3.

- The CARs were sparged with nitrogen gas to remove oxygen from the system, and then operated with continuous leachate recirculation.

- The CARs were continuously monitored for biogas, leachate and settlement over periods of 286 and 202 days for the UK MBT waste and German MBT waste respectively.

*24 hours is shown later as the time required for primary consolidation to be complete

- The hydraulic conductivity of the degraded waste at a vertical stress of 50 kPa was determined before the load increase to 150 kPa (unfortunately, this step was omitted for the UK MBT waste).

- The load was then increased to 150 kPa and the CARs were monitored for a further 61 and 77 days for the UK German MBT wastes respectively.

- The final hydraulic properties (k and n_d) and other parameters n, ρ_d and ρ were determined at 150 kPa, at the end of each test.

3.5.1.1 Determination of drainable porosity

The reactor was filled with synthetic leachate to saturate the waste from the base to the top using a peristaltic pump. This was undertaken in stages to saturate only a small proportion of the waste at a time until complete saturation of the waste was achieved. Leachate levels within the piezometers up the side of the reactor were allowed to stabilise before a further volume of synthetic leachate was added. After the waste had been fully saturated*, the total porosity of the waste was determined. Then a drainage procedure was carried out on the waste inside the reactor, lasting a period of several hours. The waste was allowed to drain in stages under conditions of free downward gravity. At each stage, small volumes of leachate were drained and the resulting change in hydrostatic head in the waste was measured.

*waste was allowed to saturate overnight and saturation was evidenced by stabilised levels in piezometers.

The leachate was drained through the lower outlet and the piezometers on the side of the reactor were used to measure the piezometric head. Piezometric levels were allowed to stabilise before readings were taken. The hydrostatic leachate head (h) indicating the level to which the waste in the reactor had become saturated was plotted against the cumulative volume (V) of leachate drained. The slope of the line (S) through these data points at any particular elevation within the waste is directly related to the drainable porosity.

n_d = Δ volume of leachate drained/ Δ volume of waste saturated

$$n_d = \frac{\Delta V \times 100\%}{\left(\Delta h.\pi r^2\right)} = \frac{1 \times 100\%}{S.\pi r^2} \qquad \text{Eq. 3.16}$$

where, r = radius of CAR and S = slope of the graph = Δh/ ΔV

The drainable porosity of the MBT waste in the CARs was determined at the initial stage (no applied load), at 50 kPa, and at 150 kPa after the experiments on biodegradation and settlement behaviour were ended.

3.5.1.2 Determination of saturated hydraulic conductivity

Following saturation of the waste, leachate was pumped to the top of the CAR using a calibrated peristaltic pump. The leachate inflow was increased steadily until the outflow rate through the lower outlet matched the inflow. The test was run in downward flow direction. An arrangement was made to the CAR to allow a saturated hydraulic conductivity test to be undertaken. Leachate flow from the outlet port was conveyed through a tube connected to a lower end of an inverted T-connector, fastened to a vertical stand whose

elevation could be easily altered. The other lower end of the T-connector was connected to a tube that drained leachate to a container (Fig. 3.8). The top end of the T-connector was left open to atmosphere to prevent siphoning effects. Flow was adjusted by raising or lowering the T-connector. The outflow volume and the head at piezometer ports within the waste (at 250 mm and 400 mm above the reactor base) were measured every 15 minutes. The steady state condition was established when the inflow matched the outflow and the leachate levels in the piezometers were stable. The procedure followed is based on a previous study on MSW (Beaven et al., 2005). The hydraulic conductivity for this constant head flow test was obtained using the Darcy's equation:

$$k=q/iA \qquad \text{Eq. 3.17}$$

where, q is the flow rate, i is the hydraulic gradient, and A is the cross sectional area. The hydraulic gradient in the waste was determined from the readings of piezometric heads as:

$$i= \Delta h/L \qquad \text{Eq. 3.18}$$

where, Δh is the loss of hydraulic head over the distance L separating the two piezometers

The hydraulic conductivity of the waste in CARs was determined at the initial stage (no applied load), at 50 kPa (before and after degradation), and at 150 kPa after the experiments on biodegradation and settlement behaviour were ended.

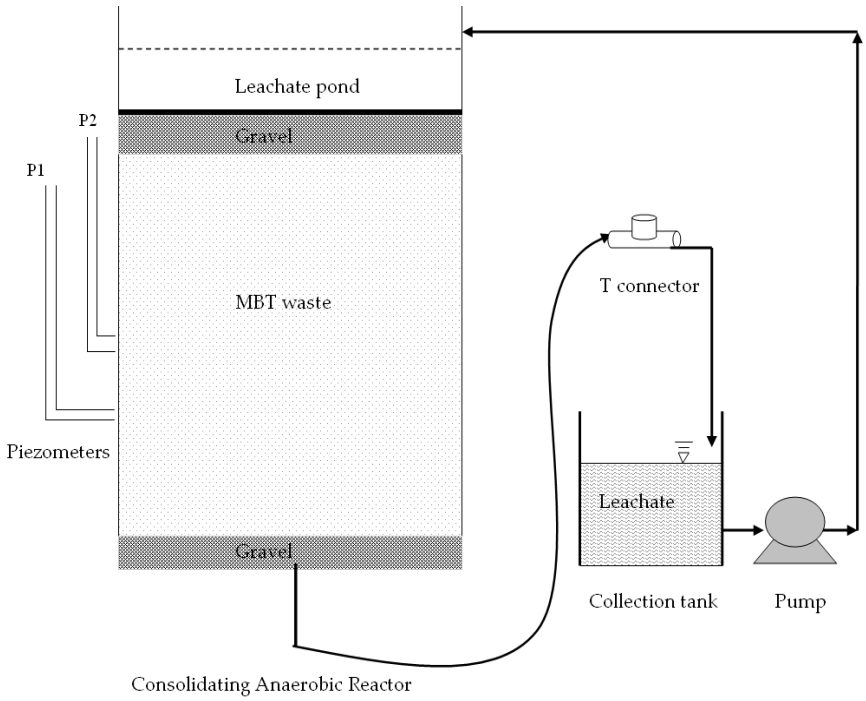


Figure 3.8: Experimental set up for the measurement of saturated hydraulic conductivity

3.5.1.3 CAR start up and operation

The long term biodegradation and settlement behaviour of each MBT waste was investigated in a control reactor (CAR1) and a test reactor (CAR2) (see Figure 3.9). After the measurement of drainable porosity and hydraulic conductivity at 50 kPa, the CARs were drained and completely sealed with appropriate valves for gas and leachate handling. The reactors were then sparged with nitrogen gas to remove any oxygen trapped within the reactor.

Figure 3.9: Consolidating anaerobic reactors, CAR1 and CAR2

CAR2 was filled with 80 litres of leachate comprising an inoculum of anaerobically digested sewage sludge (10% vol.) derived from an anaerobic digester at Millbrook Sewage Works (Southern Water, UK). This was done to ensure the presence of viable methanogenic bacteria and to accelerate the initiation of methanogenesis. The addition of sewage sludge (10% vol.) to the waste sample was found to be the optimal way of ensuring rapid anaerobic degradation (Ivanova, 2007).

In the control reactor (CAR1), 80 litres of leachate mixed with acetic and propionic acids at a concentration of 10 grams per litre each was added to suppress microbial activity and prevent the onset of methanogenesis. Inhibition by organic acids accumulation has been demonstrated in numerous studies (Angelidaki et al., 1993; Gachet et al., 2003; Bayard et al.,

2005; Ivanova et al., 2008*a*). The leachate and waste in CARs were sparged again with nitrogen gas to remove any trace of oxygen from the system and the leachate was recirculated continuously at a flow rate of 1.38 litres per hour from the bottom to the top of each reactor using a peristaltic pump (Watson Marlow model 323, USA). CAR1 was operated at ~20°C in a controlled temperature room and CAR2 was maintained at a constant mesophilic temperature of 30°C using a heat blanket to establish a favourable growth environment for the microorganisms in the reactor (Figs. 3.10 & 3.11).

The operation of the CARs in this way was designed to isolate settlements due to mechanical creep from those due to biodegradation, by comparing the settlements measured in each. The CARs were operated for 347 and 279 days for the UK MBT and German MBT wastes respectively. The load was maintained at 50 kPa for the first 286 and 202 days of the experiment on the UK and German MBT wastes respectively. Thereafter, load was increased to 150 kPa for the remainder of the test in order to investigate any dependence of the creep characteristics of the waste on the applied load (Figs. 3.12 & 3.13).

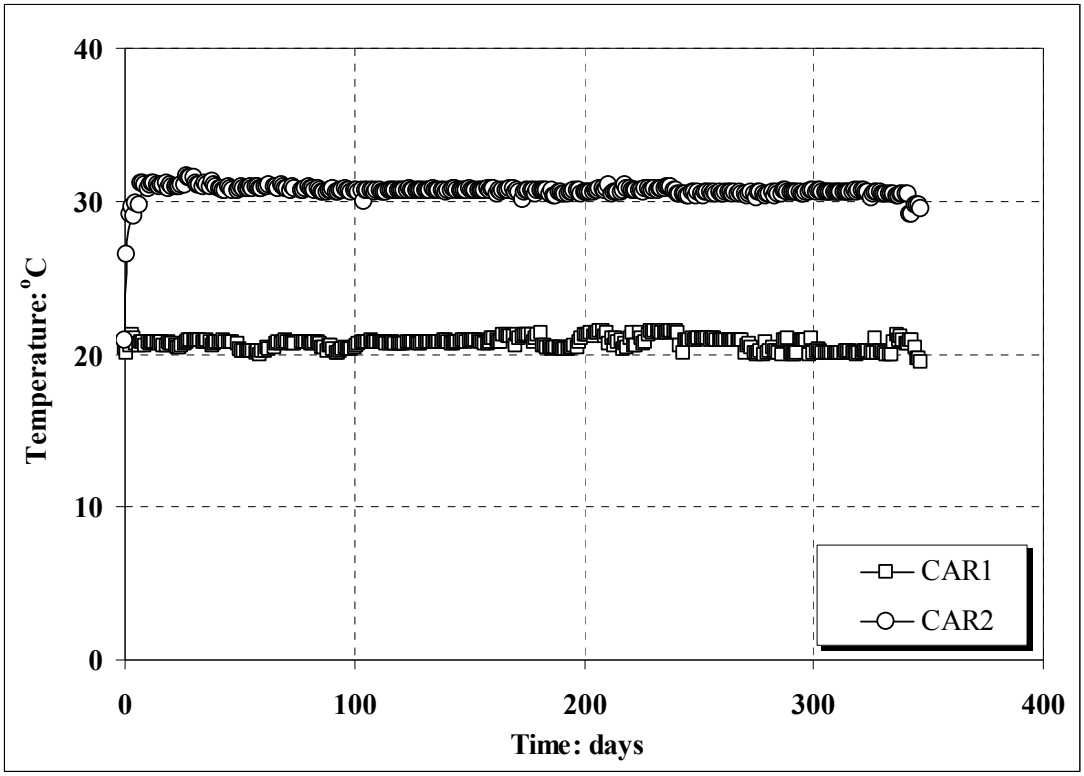


Figure 3.10: Monitored temperature of CAR1 and CAR2 (UK MBT waste)

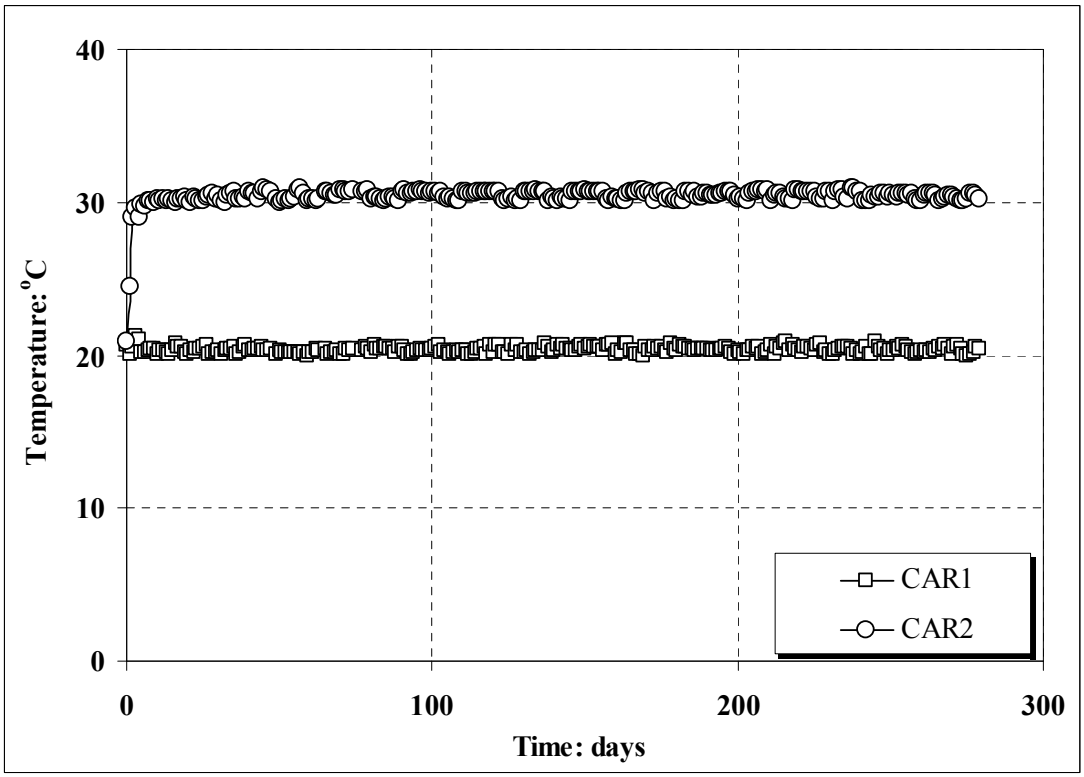


Figure 3.11: Monitored temperature of CAR1 and CAR2 (German MBT waste)

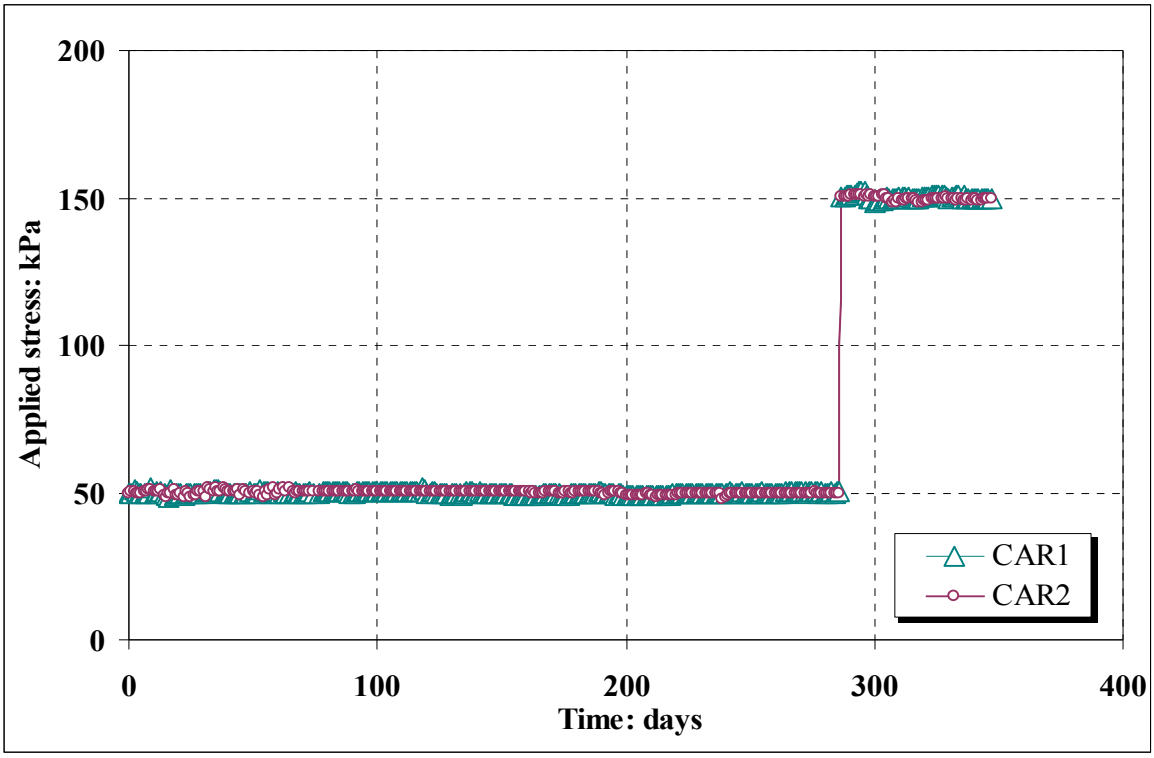


Figure 3.12: Monitored applied load to CAR1 and CAR2 (UK MBT waste)

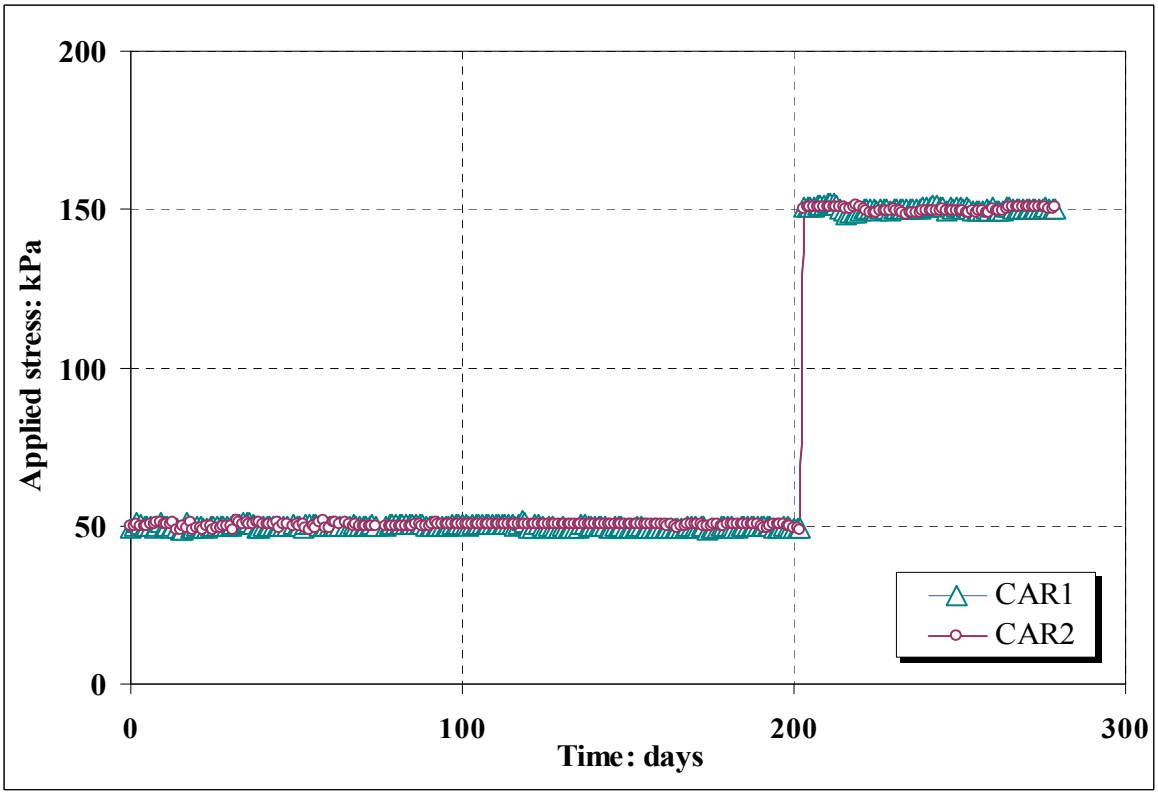


Figure 3.13: Monitored applied load to CAR1 and CAR2 (German MBT waste)

3.5.1.4 Monitoring and testing programme

Biogas volume and composition, leachate quality and waste settlement were monitored on a regular basis to understand the degree of waste stabilisation. Leachate samples were collected from each CAR every three days during the first three months of operation and weekly thereafter. These samples were analysed for pH, electrical conductivity (EC), ORP (redox potential), total organic carbon (TOC), dissolved organic carbon (DOC), inorganic carbon (IC), volatile fatty acids (VFAs), total nitrogen (TN), ammoniacal nitrogen (NH_4-N), heavy metals, chloride, calcium and magnesium ions. Leachate samples from the CARs were analysed immediately for pH, EC and ORP, and then preserved and stored at 4°C prior to analysis for other parameters. The volume of leachate withdrawn from the reactors at each sampling was replaced by an equal volume of fresh synthetic leachate. The volume of biogas produced was recorded daily and analysed for gas composition.

3.5.2 Biochemical Methane Potential Reactors

The biochemical methane potential (BMP) reactors were run in parallel with the CARs to assess the biodegradability of each MBT waste under anaerobic conditions by measuring the production of biogas. The anaerobic biodegradation potential of the waste was characterised in terms of the biogas yield, solids composition (LOI, TC, cellulose, hemicellulose and lignin contents) and assessment of leachate characteristics at various stages of the biodegradation process.

To accelerate degradation of the waste by anaerobic bacteria, 500 ml of synthetic leachate containing anerobically digested sewage sludge seed (10%

by vol.) was added to each reactor. The synthetic leachate had previously been sparged with nitrogen gas to remove oxygen. Two further control reactors (B13 and B14) were also prepared (containing only 500 ml synthetic leachate and inoculum) in order to measure the biogas produced from the sewage sludge alone. After mixing the waste with synthetic leachate containing sewage seed, all the reactors were flushed with nitrogen gas and then kept and sealed under a nitrogen atmosphere in an anaerobic cabinet before being placed in a water bath at 30°C to promote mesophilic methanogenic conditions. No mechanical mixing of the waste was carried out during the test.

Individual reactors were sacrificed sequentially during the test to observe solids compositional changes and leachate characteristics at different stages of degradation. For the UK MBT waste, the reactors were sequentially terminated and opened for sampling on day 9 (B1), 23 (B2), 37 (B3), 45 (B4), 53 (B5), 60 (B6), 71 (B7), 87 (B8), 128 (B9) 164 (B10), 249 (B11) and 347 (B12) to allow leachate and solid compositional changes to be tracked. For the German MBT waste, the reactors were terminated and sampled on day 10 (B1), 21 (B2), 30 (B3), 41 (B4), 52 (B5), 60 (B6), 71 (B7), 90 (B8), 112 (B9) 152 (B10), 207 (B11) and 279 (B12).

Each waste sample was analysed for total carbon, total nitrogen, LOI, cellulose, hemicellulose and lignin, and the leachate samples were analysed for pH, TOC, IC, VFAs, TN, NH_4-N, chloride, calcium and magnesium. Biogas samples taken from each BMP reactor were analysed for methane (CH_4) and carbon dioxide (CO_2) using gas chromatography technique.

3.6 Analytical methods

Different analytical procedures used to analyse biogas, solid waste and leachate samples are explained below.

3.6.1 Biogas analysis

The biogas composition (methane and carbon dioxide) in the CARs was measured daily as a percentage by volume using an infra-red gas analyser GASCARD II Plus. This analyser comprises a robust infra-red source, a gas sampling cell, two infra-red filters and a matched twin element detector. The infra-red absorption wavelength enables the detection of the gas and the strength of the absorption gives a measure of the gas concentration. This analyser was calibrated by the manufacturer when it was purchased and the calibration verified using standard gases of known methane and carbon dioxide content.

The biogas accumulated in the head space of BMP reactors was collected in a gas tight syringe and analysed immediately. The composition of the biogas (methane and carbon dioxide) was determined by gas chromatography in Varian CP3800 GC (Varian, USA) fitted with a HaySep C column and a molecular sieve 13x (80-100 mesh) operated at a temperature of 50°C. Argon was used as a carrier gas at a flow of 50 ml/minute.

3.6.2 Solid waste analysis

Waste solids were analysed for LOI, total carbon (TC), total nitrogen (TN), cellulose (C), hemicellulose (H) and lignin (L). For LOI, TC and TN analyses all non grindables were removed from the sample and remaining

components were analysed according to the procedure described in Sections 3.2.5 and 3.2.6. Plastic and non grindable components were removed from the sample and the remainder analysed for lignin, cellulose and hemicellulose as detailed in the Section 3.2.7. All solids analyses were performed in triplicates and average values are reported (the maximum variation from the mean was 5%).

In fibre analysis, plastics (if present in the waste sample) are likely to be characterized as lignin and this may give higher concentrations for the lignin content (Barlaz, 2006; Kim, 2004). This may lead to the erroneous measurement of both lignin and (C+H)/L ratio. Prior to fibre analysis, small plastic fragments were removed from the waste samples in the particle size range greater than 1mm.

About 20 g of the unidentified fraction of the UK and German MBT waste falling in the particle size range 1mm or smaller was examined under a Stereo microscope binocular model EMT-3/10x magnification (Meiji Techno UK Ltd.), and no plastics were seen.

Assuming 5% plastics in this unidentified fraction (comprising less than 20% of the total waste sample), lignin measurements will be about 1% greater than the actual amounts giving (C+H)/L ratio of about 99% of the actual value. This shows that plastics interference, if any, is not significant.

3.6.3 Leachate analysis

All the leachate samples taken from the reactors were analysed in duplicate and the results presented are the average of these measurements.

3.6.3.1 pH, Electrical Conductivity and Oxidation Reduction Potential

The pH, electrical conductivity (EC) and oxidation reduction potential (ORP) of leachate samples collected from CARs were analysed immediately. A Jenway Model 3010 digital pH meter (Jenway, UK) was used to measure the pH and was calibrated using buffers. EC and ORP were analysed using Hanna Instruments portable meters models HI 99301 and HI 8424 respectively. The calibrations of these instruments were checked using standard solutions.

3.6.3.2 Total Organic Carbon, Dissolved Organic Carbon and Inorganic Carbon

Total organic carbon (TOC), dissolved organic carbon (DOC) and inorganic carbon (IC) analyses were carried out using a high temperature TOC analyser Dohrmann-Rosemount DC 190, USA. This analyser contains a vertical quartz combustion tube packed with cobalt catalyst. The furnace is operated at 600°C and oxygen flows through it at a rate of 180-220 ml/minute. The sample is vaporised and the carbon oxidised to CO_2 is measured by infra-red detector. A sample of 5 ml for TOC and DOC determination was preserved with 50µl of 100% HCl and stored in plastic vials in refrigerator at 4°C. The DOC sample was filtered through a micro-glass fiber filter paper (Fisher Scientific, UK).

3.6.3.3 Volatile Fatty Acids

Volatile fatty acids (VFA) in samples were quantified by gas chromatography using a Shimadzu-2010 GC instrument, which uses a capillary column SGE BP 21 and helium as the carrier gas. The flow of this gas in the column was 1.86 ml/minute. The temperature in the GC increased from 60°C to 210°C over 15 minutes with a holding time of 3 minutes; the temperatures in the injector and the detector were 100°C and 250°C respectively. In each VFA run, three standard solutions containing 50, 250 and 500 mg/litre of acetic, propionic, iso-butyric, n-butyric, iso-valeric, n-valeric, hexanoic and heptanoic acids were used for the calibration. The GC was attached to an auto sampler (Shimadzu AOC-20S) and an auto injection unit (Shimadzu AOC-20i). A 5 ml sample was preserved with 10% formic acid and stored in plastic vials at 4°C

3.6.3.4 Ammoniacal Nitrogen and Total Nitrogen

Ammoniacal nitrogen was measured by steam distillation using a Foss Tecator Kjeltec System 1002 distillation unit (Foss Tecator, Sweden) immediately after sampling, in accordance with Standards Methods (APHA, 2005). Total nitrogen was analysed using TOC analyser, Dohrmann-Rosemount DC 190, USA equipped with a Dohrmann ozonator.

3.6.3.5 Heavy metals

Heavy metals (copper, zinc, nickel, cadmium, chromium and lead) in the samples were analysed using a Varian Spectra AA-200 atomic absorption spectrophotometer (AAS) (Varian, Australia). The AAS is based on the

absorption of UV or visible light by gaseous atoms. The heavy metals in sample are measured when they are atomised by an air-acetylene flame; the atoms in the flame then absorb the wavelength that is sent by the light source: a hollow cathode lamp. Each metal requires a specific lamp designed to emit the required wavelength. The AAS machine was switched on 20 minutes before the injection of the sample to allow the equipment to stabilise and the lamp to warm up. At the start of each analysis a calibration for each metal was prepared by running 5 standards. Once a linear calibration had been obtained, replicate samples (appropriately diluted where necessary) were injected.

3.6.3.6 Chloride

The chloride content in the leachate was measured using the Visocolor ECO test kits and a photometer PF-11 (Marcherey-Nagel, Germany). These kits are designed for use in the field and are based on a colorimetric measurement scale. The reaction of mercuric thiocyanate with chloride ions produces mercuric chloride and liberates thiocyanate ions which give an orange colour in the presence of ferric salts.

3.6.3.7 Calcium and Magnesium

The calcium and magnesium ions in leachate samples were analysed with flame atomic absorption spectrometry using Varian Spectra AA-200 (Varian, Australia).

3.6.4 Limits of detection and accuracy of analytical methods

For the analytical methods used the limits of detection and accuracy are given in Table 3.5.

Table 3.5: Instrument limits of detection and accuracy

Instrument	Detection limit	Accuracy
TOC analyser, Dohrmann-Rosemount DC 190	0.2 mg/litre	± 2%
GC Shimadzu-2010 (VFA analysis)	1 mg/litre	± 2.5%
AAS Varian Spectra AA-200	0.01 mg/litre	± 2%
Ammoniacal nitrogen , Kjeltec System 1002 distillation unit	1 mg/litre	± 5%
Chloride Visocolor ECO test kits and photometer PF-11	10 mg/litre	± 5%
Infra red gas analyser, GASCARD II Plus	0.02%	±2.5%
GC Varian CP3800 (Biogas analysis)	0.1%	± 2%

CHAPTER 4 RESULTS AND DISCUSSION-BIODEGRADATION

This chapter presents the results of the investigations carried out in large scale consolidating anaerobic reactors (CARs) to quantify the biodegradation of MBT wastes treated to typical UK and German standards. The gas generating potential, gassing rate and biogas composition are evaluated and compared. The changes in the composition of leachate during waste biodegradation are presented and compared in terms of pH, conductivity, redox potential, TOC, DOC, IC, TN, NH_4-N, chloride, calcium, magnesium and heavy metals. The degradability of MBT wastes is characterised by means of small scale BMP reactors in terms of the change in solids composition with the progression of decomposition. The results are compared with those obtained for raw MSW by Ivanova (2007, 2008*a*, 2008*b*).

4.1 Large scale CARs

4.1.1 Biogas production

The cumulative and daily gas production for the UK and German MBT wastes are illustrated in Figs 4.1 and 4.2. The measured volume of biogas produced was corrected to dry gas at STP using ambient temperature and pressure measurements and values of water vapour pressure as described in Section 3.3.2. Cumulative biogas production was calculated by summing the

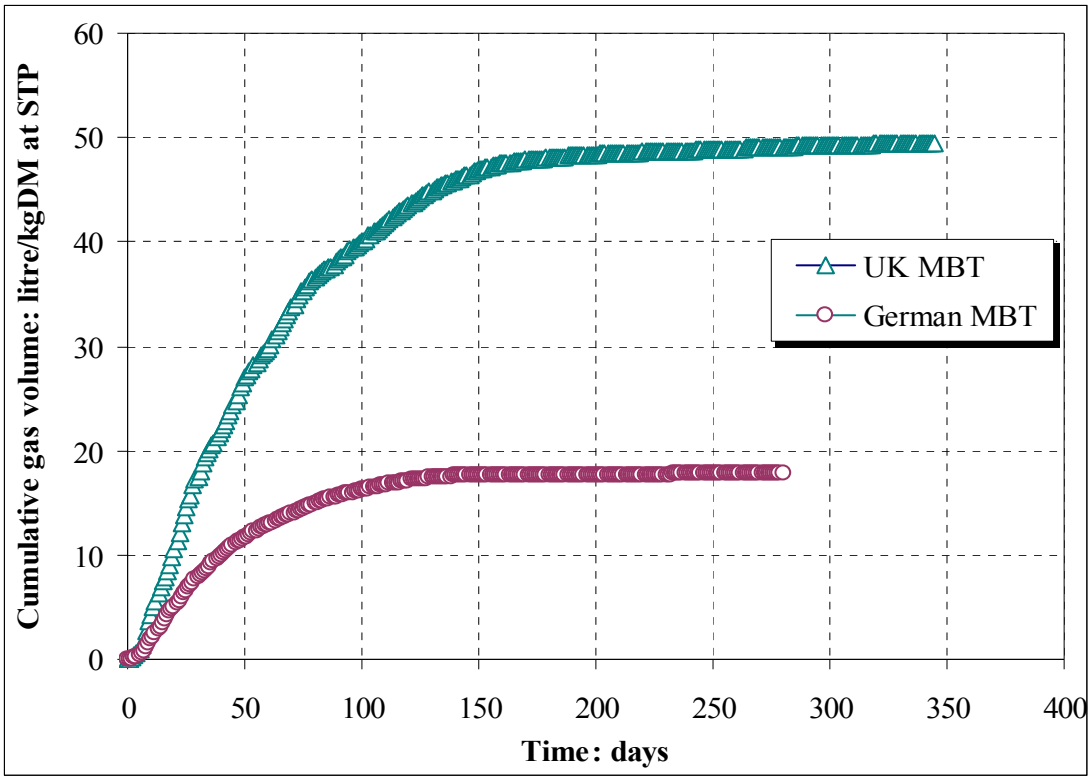
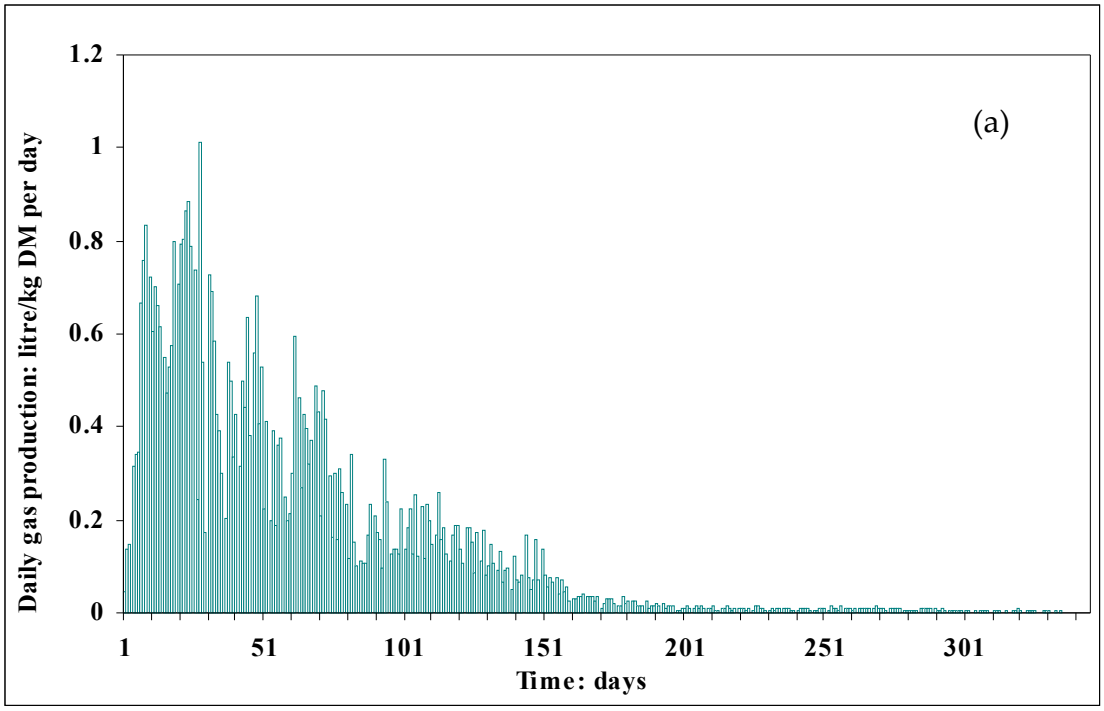


Figure 4.1: Cumulative biogas production at STP in CAR2 for the UK and German MBT wastes

daily biogas production values. Figure 4.1 shows that gas production started soon after filling the test reactor, CAR2, for both the UK and German MBT wastes. Biogas production was relatively low during the first week of operation, presumably due to the time taken for the acclimatisation of the methanogenic bacteria and the accumulation of VFA which kept the pH low (Figs. 4.8 & 4.9). The biogas production then increased and methanogenic conditions were quickly established as confirmed by the change in biogas composition to about 60% CH_4 and 35% CO_2 by volume (Figs. 4.3 & 4.4). The acidogenic phase was virtually absent; probably due to the degradation of some organic compounds during pretreatment. This is in agreement with the findings of Bayard et al. (2008) and Bockeris et al. (2003). Biogas production for both the wastes followed a similar trend. The gas production rate



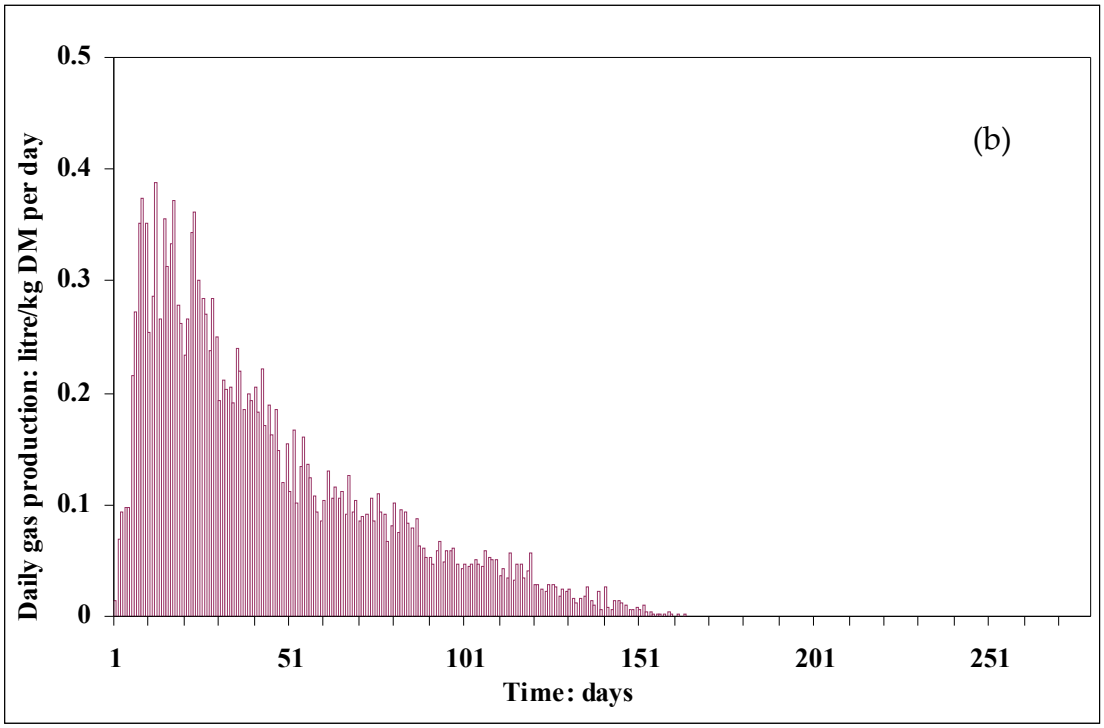


Figure 4.2: Daily biogas production in CAR2 (a) UK MBT waste (b) German MBT waste

increased to a maximum of about 0.9 litre/kg DM/day and 0.4 litre/kg DM/day followed by a gradual decline for the UK and German MBT wastes respectively (Fig. 4.2). The apparent large variation in gas generation rates in Fig. 4.2 between some successive days was due to a problem with the solenoid valve, which failed to open owing to condensate in the gas line. A new solenoid valve was installed after about 100 days in the experiment on UK MBT waste, which worked satisfactorily thereafter. The solenoid valve worked satisfactorily throughout the experiment on German MBT waste.

The problem with the solenoid valve may mean that the volume of gas calculated is an underestimate of that actually produced, if in fact the presence of the condensate caused the headspace pressure to build up to a higher value before venting. Although there was no direct evidence for this (pressure in the headspace was not monitored), an estimate of any possible under recording may be made. It is assumed that on days when the daily gas generation rate was significantly below the rates on the two days on either side, the unrecorded volume of gas is given by the difference between the value recorded and the average of the two days on either side. The problem was investigated and there was at most a 6% error.

Slight variations in gassing rates, as were apparent with the German waste even when the solenoid valve was functioning correctly (no condensate), could be related to natural vagaries in microbiological reaction rates and/or the release of previously-trapped or generated gas into the reactor's headspace in sporadic events (ebullition).

The gas generation rates in the CARs and BMP reactors are comparable for each type of MBT waste. Biogas production in the BMP reactors substantially ceased after about 100 days, giving a slightly smaller biogas potential than

CAR2. This could be due to the fact there was no mixing in the BMP reactors compared with continuous leachate recirculation in the CARs.

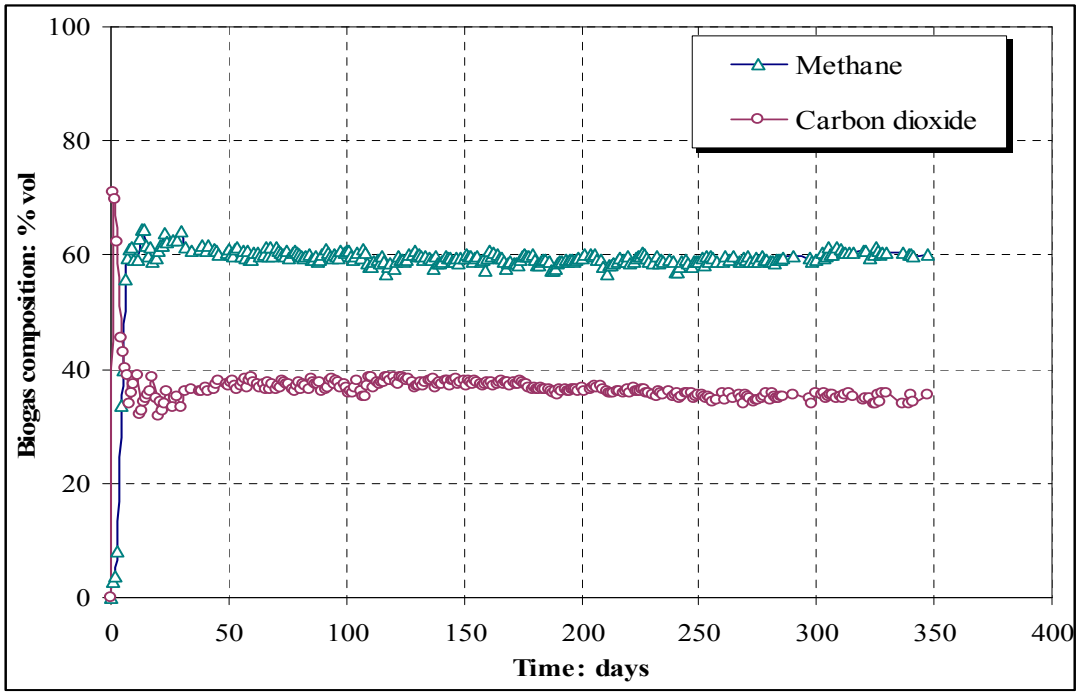


Figure 4.3: Biogas composition in CAR2 for the UK MBT waste

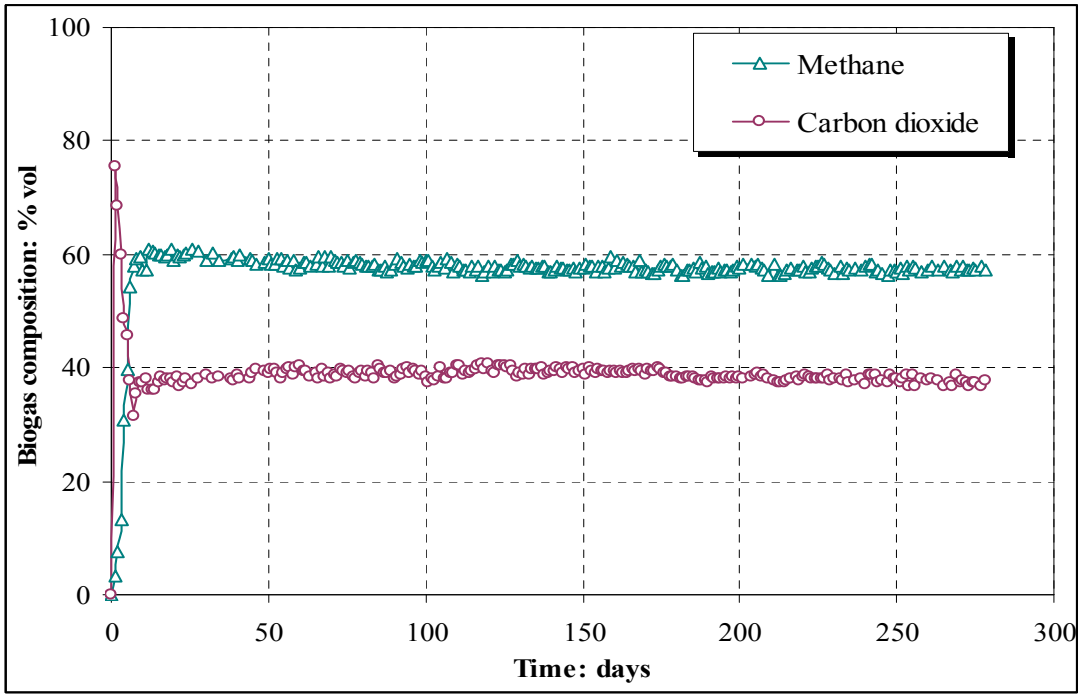


Figure 4.4: Biogas composition in CAR2 for the German MBT waste

The decrease in biogas production was consistent with the VFA depletion observed during that period (Figs. 4.8 & 4.9). Most of the gas had been liberated by day 100. The gas production rate decreased to less than 0.01 litre/kg DM/day at about day 200 for the UK MBT waste and day 150 for the German MBT waste. Thereafter, gas production continued at a much lower rate until day 280 and 195 when it had effectively ceased for the two wastes respectively. A similar observation for the gassing rate was made by Knox & Robinson (2007) regarding the continuation of low level emissions in landfills over the long term.

The gas composition in the two wastes was quite similar. The methane content of the biogas ranged from 58% to 62% and the carbon dioxide content was in the range 35-40% (Figs. 4.3 & 4.4). The gassing potential of the German MBT waste was low compared with the UK MBT waste and the total biogas yield was 49.46 litres/kg DM (UK MBT waste) and 17.74 litres/kg DM (German MBT waste). The increased gas production in the UK MBT waste could be explained by the lower degree of biological pretreatment, as can be verified from the higher values of the organic content i.e. the LOI, cellulose and TC contents in Table 3.3. The biogas yield of the UK MBT waste is higher than in some other studies (De Gioannis et al., 2009, Leikam and Stegmann, 1999) but in the range reported by Horing et al. (1999).

The control reactor, CAR1, for the UK and German MBT residues did not produce any biogas owing to the acid environment (evident from the pH) caused by acidification of the reactor with acetic and propionic acids at the start of the experiment. The absence of biogas indicates the suppression of microbial activity and the successful inhibition of biodegradation confirming

the findings of Angelidaki et al.(1993); Gachet et al.(2003); Bayard et al.(2005) and Hossain et al.(2003). Ivanova (2007) studied the biodegradation and settlement behaviour of raw MSW in CARs for 919 days. Figure 4.5 compares the cumulative biogas produced from raw MSW (Ivanova et al. 2008*a*) with the UK and German MBT wastes. The raw MSW experienced a long acidogenic phase of about 40 days before any biogas was produced: in contrast, there was no evidence of acidogenic phase in the MBT wastes. The cumulative gas volume for raw MSW was significantly higher (255.4 litres/kg DM) and the gassing rate was also much higher, in the range 2 to 8 litre/kg DM/day (Fig. 4.6). This is attributable to the higher amount of organic waste components in the raw MSW. The only similarity was in the gas composition as the methane and carbon dioxide contents of the biogas continued in the range 60 – 65% and 30 – 35% respectively (Fig. 4.7)

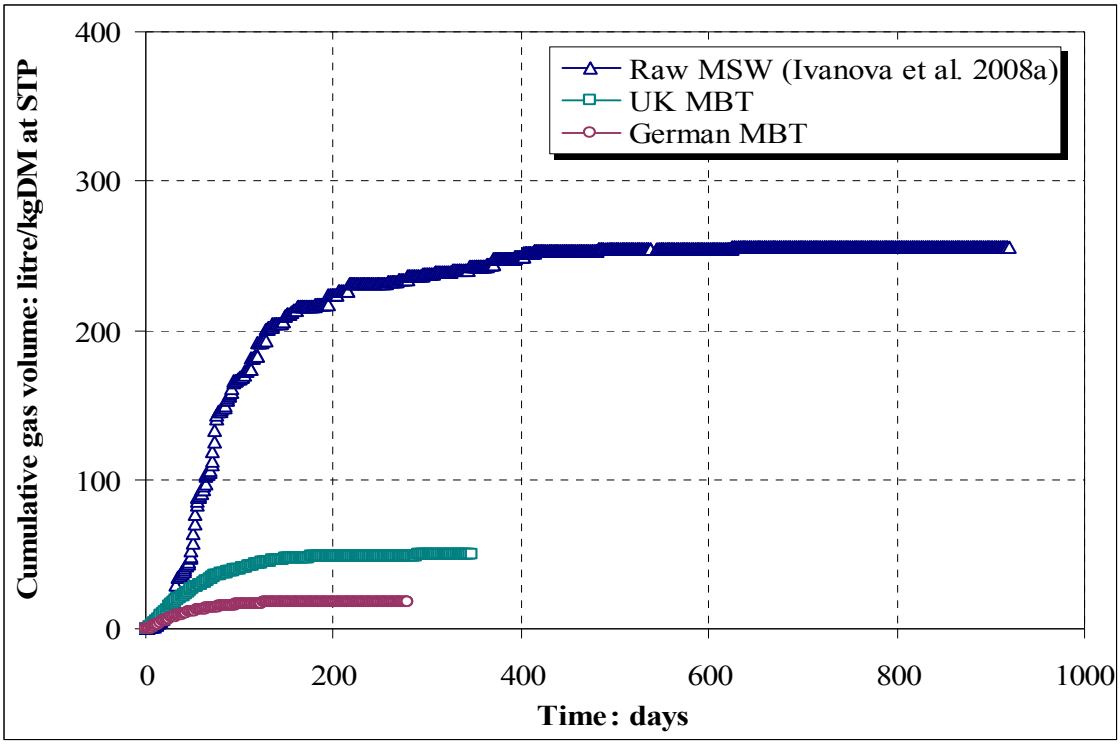


Figure 4.5: Comparison of cumulative biogas produced in raw MSW and pre-treated wastes

The comparison of results with the raw MSW revealed substantial reduction in gas generating potential and gassing rate of MBT wastes. The reduced gassing potential of the MBT wastes demonstrates diversion of the degradable fraction away from landfill as a result of the biological pretreatment, but also that landfill gas control measures will still be needed to prevent fugitive gas emissions

Thus the need to mitigate gaseous emissions will continue with MBT wastes, but the lower rates of gas production (8-20% of that for raw MSW) must be considered in the size and design of gas collection systems, and also in the economic evaluation of gas to energy projects.

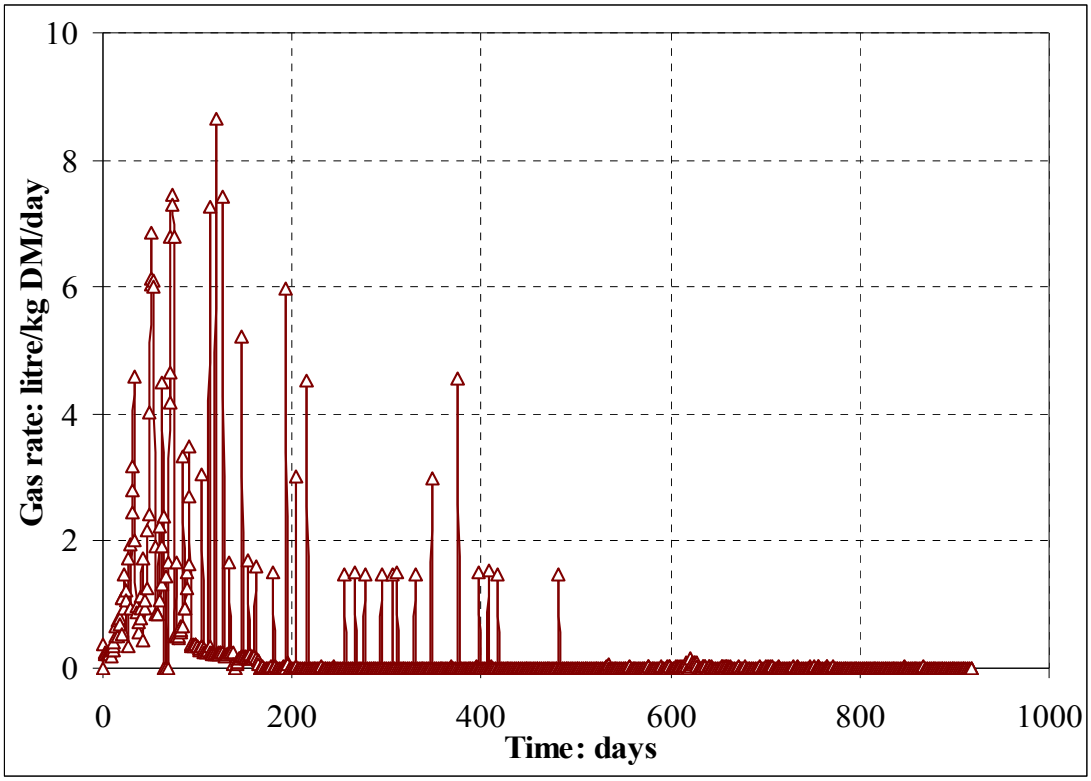


Figure 4.6: Daily biogas production in raw MSW (Ivanova et al. 2008*a*)

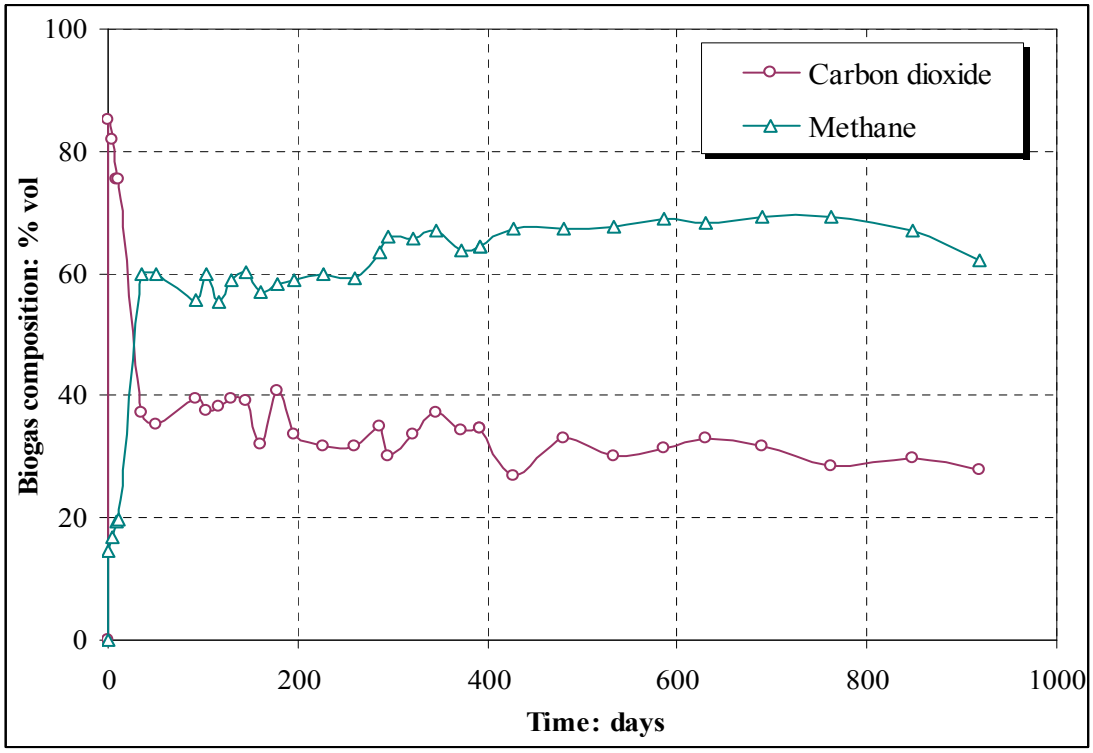


Figure 4.7: Biogas composition in raw MSW (Ivanova et al. 2008*a*)

4.1.2 Leachate composition and characteristics

Leachate quality was characterised in terms of pH, conductivity , redox potential, VFA, TOC, DOC, IC, TN, ammoniacal nitrogen, heavy metals, chloride, calcium and magnesium ions.

4.1.2.1 pH and VFA

pH is an important parameter in solid waste decomposition as methanogens producing biogas are most efficient at a pH range of 6.5-7.5 and are inhibited at lower pH values. The pH of the leachate in the test reactor (CAR2) decreased slightly due to the build up of VFA during the first week (Figs. 4.8 & 4.9). Initially, the pH of the leachate was about 7.3 which decreased slightly to about 7 and 6.9 for the UK and German MBT wastes respectively.

Apparently, the more acclimatised methanogenic bacteria coming from the sludge adapted more quickly to the environmental conditions of the reactor. This helped to transform the available organic matter into biogas faster, which led to an increase in the pH values above the neutral range; this was in line with the observed decrease of leachate VFA content during the same period. The pH values increased steadily and remained relatively stable between 7.5 and 7.7 until day 347 and 279 for the UK and German MBT waste respectively.

In the test reactor (CAR2), the total VFA concentration increased slightly in the first week due to the accumulation of hydrolytic products, and started to decrease thereafter as the methanogens utilised them as substrate for conversion to biogas. VFA do not accumulate but are quickly metabolised to

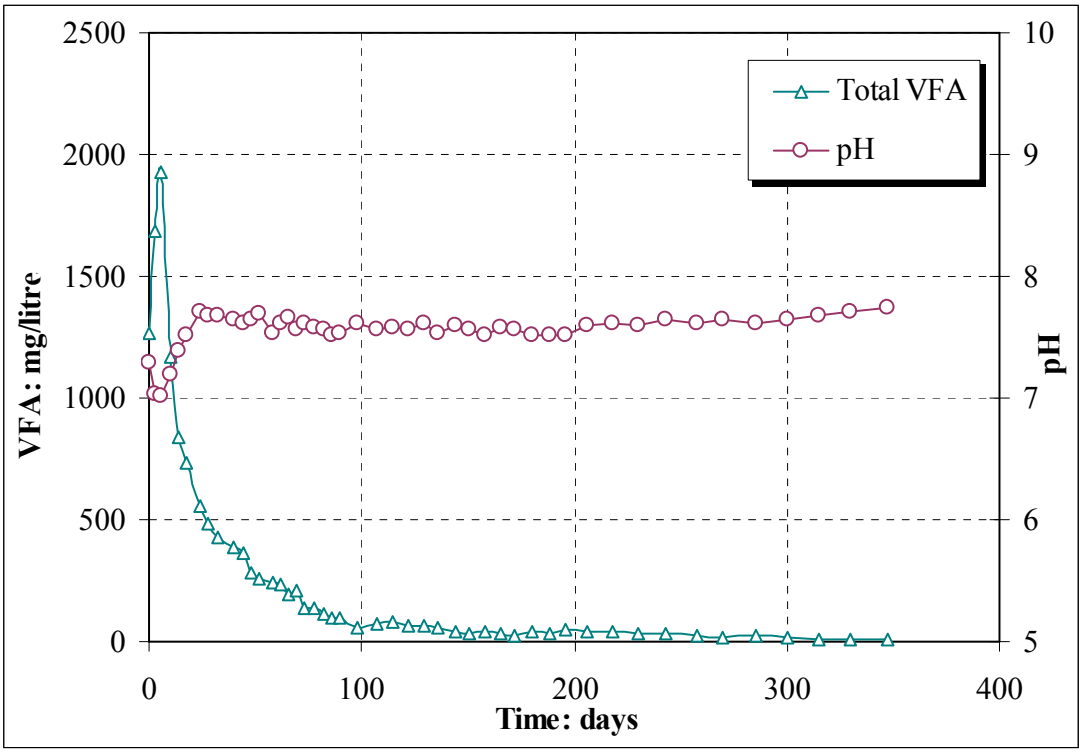


Figure 4.8: Leachate VFA and pH in CAR2 for the UK MBT waste

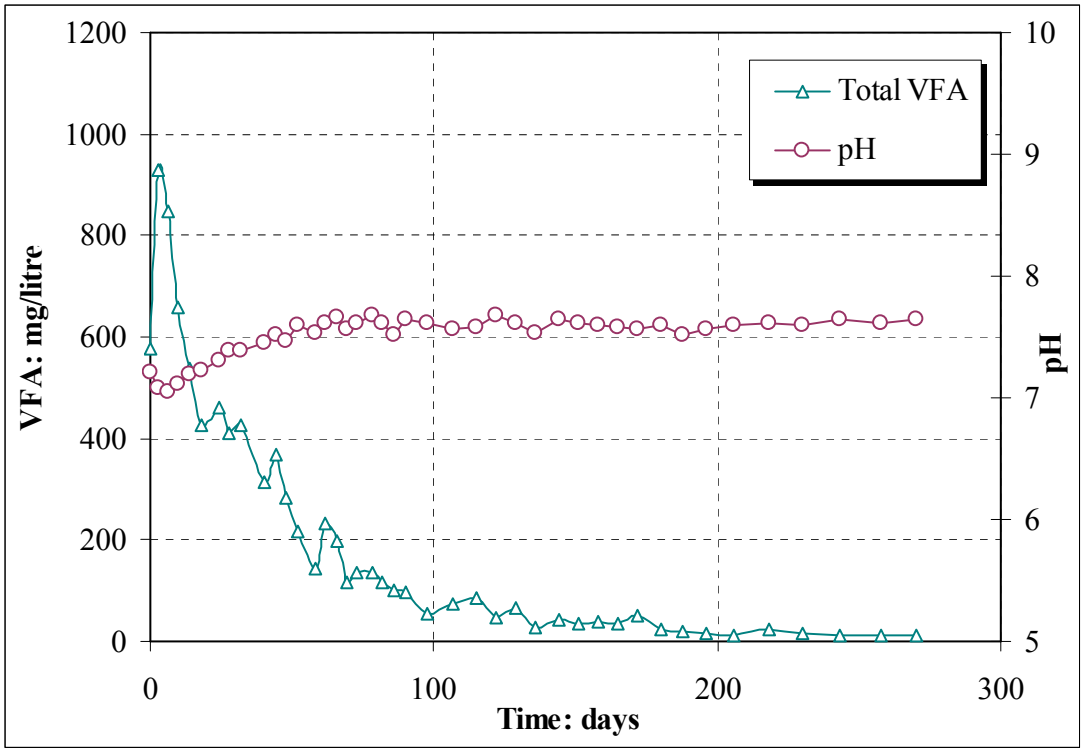


Figure 4.9: Leachate VFA and pH in CAR2 for the German MBT waste

methane and carbon dioxide. The depletion of VFA is consistent with the increase in pH of the leachate during the same period. This is in agreement with the previous findings of Agdag & Sponza (2005), and Warith (2002). The low VFA concentration is an indication that most of the available organic matter has been converted into biogas and that biological stabilisation has been achieved. The initial higher values of VFA in the UK MBT waste as compared to the German MBT waste was possibly the result of higher organic content in the UK MBT waste.

In contrast to CAR2, low pH values and higher VFA concentrations in the control reactor (CAR1) reflect the fact that the reactor had been acidified to inhibit the establishment of methanogenic conditions (Figure. 4.10).

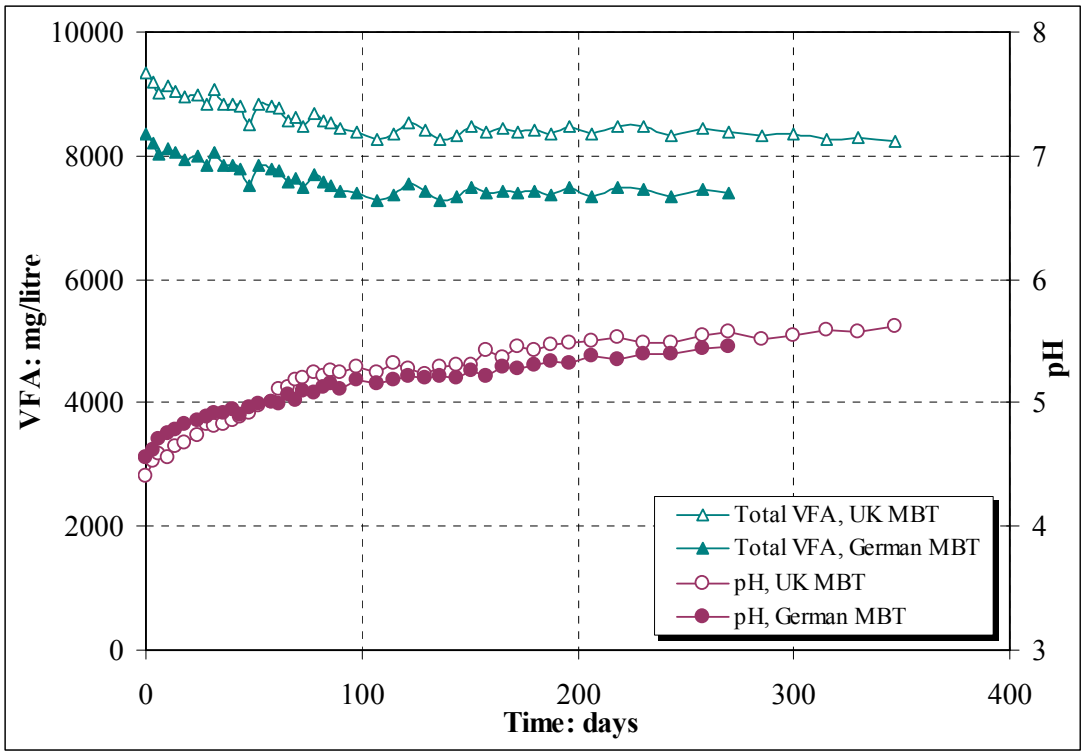


Figure 4.10: Leachate VFA and pH in CAR1 for the UK & German MBT waste

4.1.2.2 Electrical conductivity

Electrical conductivity (EC) measures the ionic strength of the leachate and is proportional to the dissolved solids concentration. The variation of leachate conductivity in the CARs for the UK and German MBT wastes is illustrated in Figure 4.11. The conductivity increase in the first few weeks may be attributed to the degradation of organics and release of the inorganic salts into the leachate. The EC of the leachate then decreased gradually and remained stable at values close to 13 and 11 mS/cm for the UK and German MBT wastes respectively. The sorption, precipitation and removal of some of the ions from the leachate may be the reason for the decrease in conductivity of leachate. The conductivity of leachate is within the range (10-20 mS/cm) cited by Robison et al. (2005).

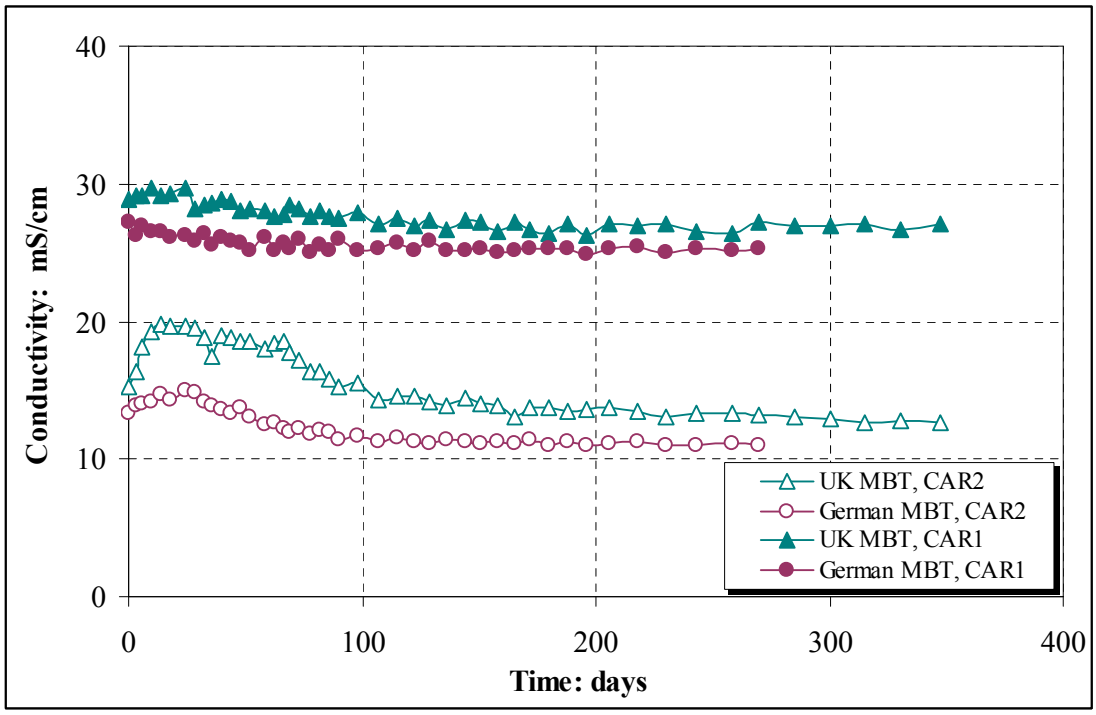


Figure 4.11: Conductivity of the leachate in CARs

In CAR1, conductivity values remained stable within a higher range throughout the experiments on the UK and German MBT wastes. The elevated value was due to the addition of acids at the start of the test. The EC did not change much because there was no activity for the waste to degrade and for precipitation, sorption or removal of ions.

4.1.2.3 Oxidation reduction potential

Oxidation reduction (redox) potential is a measure of a system's ability to facilitate oxidation or reduction reactions. The redox potential reflects the oxidation state and chemical form, influencing the solubility and sorption of many constituents in the leachate (Lu et al., 1985). The changes in redox potential in the CARs for the UK and German MBT wastes are shown in Figure 4.12. The fact that reducing conditions were developed early on is

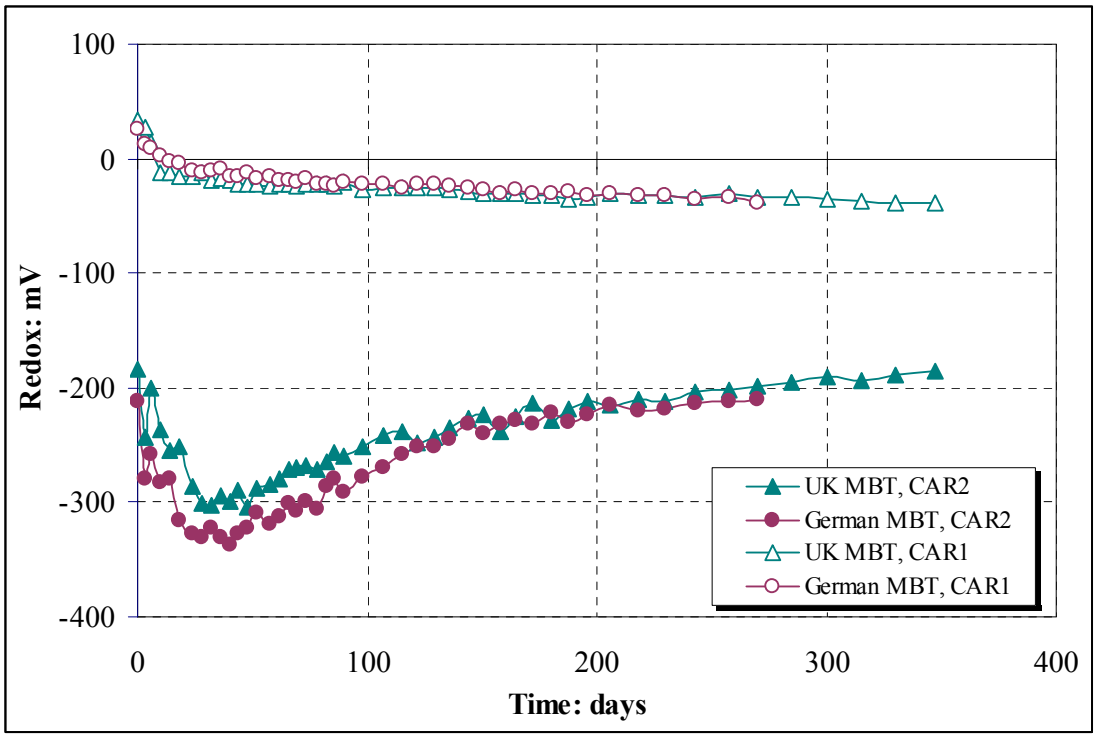


Figure 4.12: Redox potential of the leachate in CARs

shown by a sharp decrease in redox potential between 0-30 days. The redox potential reached about -300 mV which is the optimum condition for methanogenesis (Christensen and Kjeldsen, 1989; Sormunen et al., 2008; Bilgili et al., 2007). The redox potential measurements show that the leachate was strongly reducing.

In CAR1, the redox potential of the leachate for the UK and German MBT wastes was consistently high showing no real sign of degradation.

4.1.2.4 Total organic carbon, dissolved organic carbon and inorganic carbon

The organic strength of the leachate was analysed in terms of TOC and DOC and the results for CAR2 for the UK and German MBT wastes are presented in Figs. 4.13 & 4.14 respectively. The TOC concentration increased during the

first week due to the rapid release and hydrolysis of organics from the waste into the leachate at the initial stage. After the onset of methanogenic conditions (confirmed by gas composition analyses), the TOC concentration began to decrease slowly in accordance with the progression of microbially mediated stabilisation processes and confirmed by the increase in gas production and the high methane content of the biogas. The TOC concentration had become stable towards the end of the study at about 650 and 290 mg/litre for the UK and German MBT wastes respectively. This TOC is mostly recalcitrant or hardly biodegradable carbon present in the reactor - presumably lignin, and humic and fulvic acids as suggested by Kjeldsen et al. (2002). The TOC trend demonstrates that waste stabilisation was achieved.

Dissolved organic carbon concentrations in the leachate followed the same trend as the TOC indicating that the carbon in the leachate was mostly in dissolved form as the TOC values were very close to the DOC values. The initial increase in the inorganic carbon (IC) concentration of leachate is probably attributable to the conversion of carbon to inorganic forms such as carbonates. The subsequent decrease in IC is presumably due to the precipitation of carbonates of calcium and magnesium.

The lower organic content of the leachate for MBT waste is in agreement with the findings reported by Robinson et al. (2005), Kuehle-Weidemeier (2004) and Horing et al. (1999) but the values are in excess of those observed by Leikam et al. (1999) and van Praagh et al. (2009).

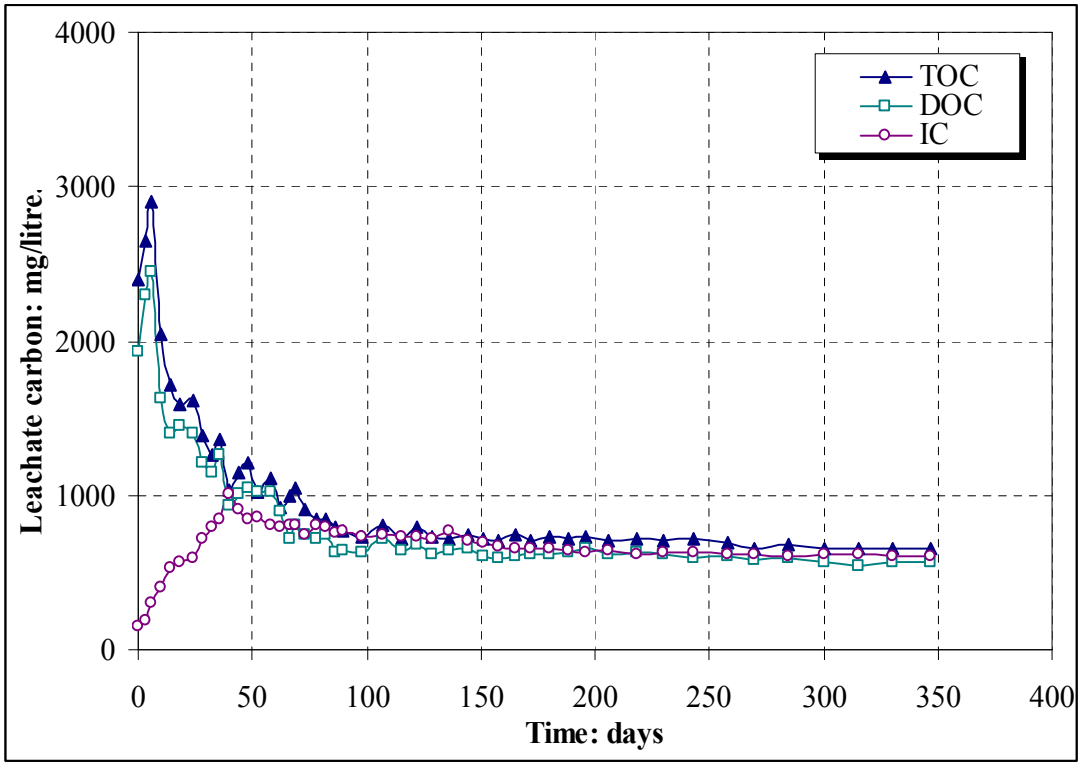


Figure 4.13: Leachate carbon in CAR2 for the UK MBT waste

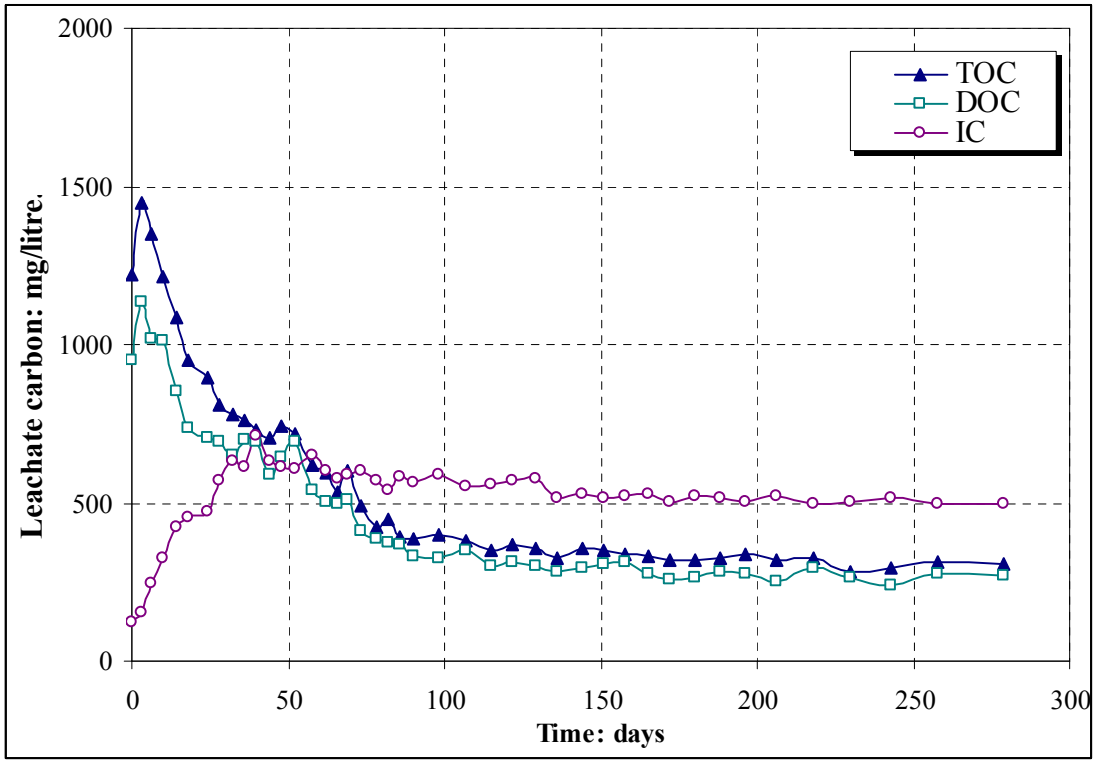


Figure 4.14: Leachate carbon in CAR2 for the German MBT waste

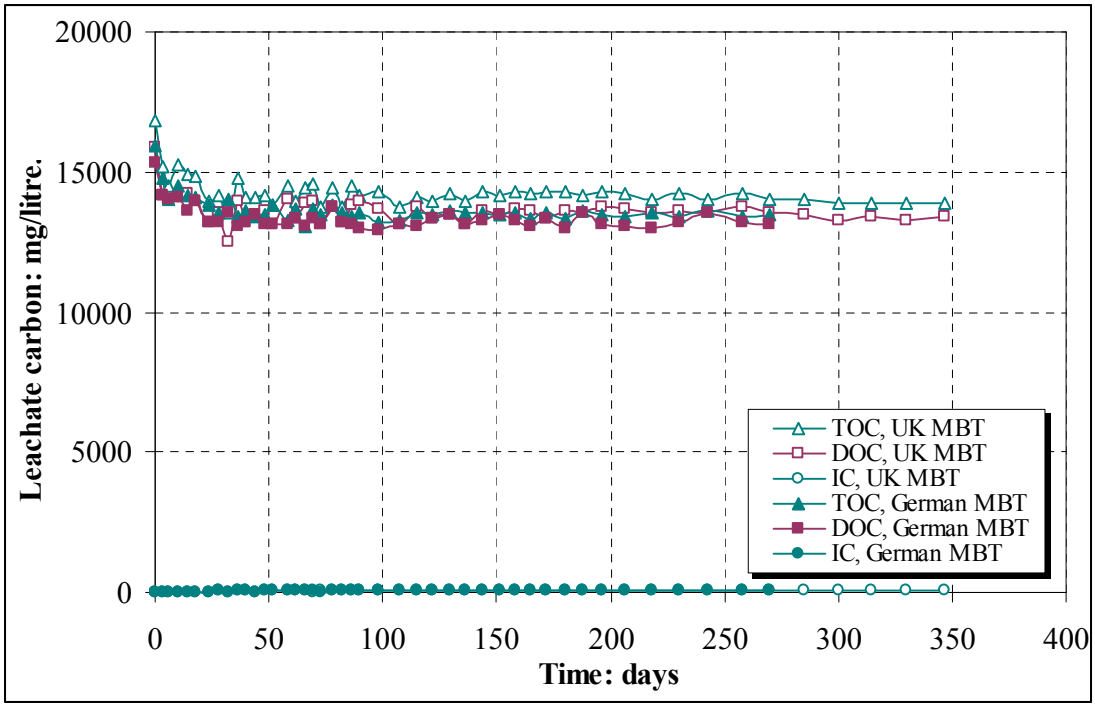


Figure 4.15: Leachate carbon in CAR1 for the UK & German MBT wastes

The organic strength i.e. TOC of the leachate from the German MBT waste was low compared with that from the UK MBT waste, consistent with the lesser organic content of the German waste owing to the different biological processing steps during pretreatment. The German MBT waste was produced after a longer duration (about 9 weeks) of anaerobic and aerobic biological treatment of raw MSW compared with 6 weeks of aerobic biological treatment for the UK MBT waste.

For both the UK and the German MBT , TOC and DOC values for the leachate in CAR1 were much higher and stable due to the acidification of the reactor with acetic and propionic acids at the beginning of the experiment (Figure 4.15).

Figure 4.16 compares the leachate load of TOC in mg/kg DM for the results obtained from the current study on MBT wastes with the data published by Ivanova et al. (2008*a*) for the raw MSW. A direct comparison of TOC in mg/litre is inappropriate owing to the difference in liquid to solid ratio (L/S) adopted in the two studies. Ivanova et al. (2008*a*) studied the leaching behaviour of raw MSW at an L/S ratio of 3.33 whereas an L/S ratio of 2 was adopted in the present study. The results of leachate load of TOC in mg/kg DM form a better basis of comparison between the raw and pre-treated wastes, and clearly demonstrate the effectiveness of pretreatment in reducing the TOC concentration in the leachate. The leachate load of carbon was the highest in raw MSW and lowest in German MBT waste. Thus, organic carbon became less leachable after pretreatment resulting in a reduction of the landfill's polluting potential.

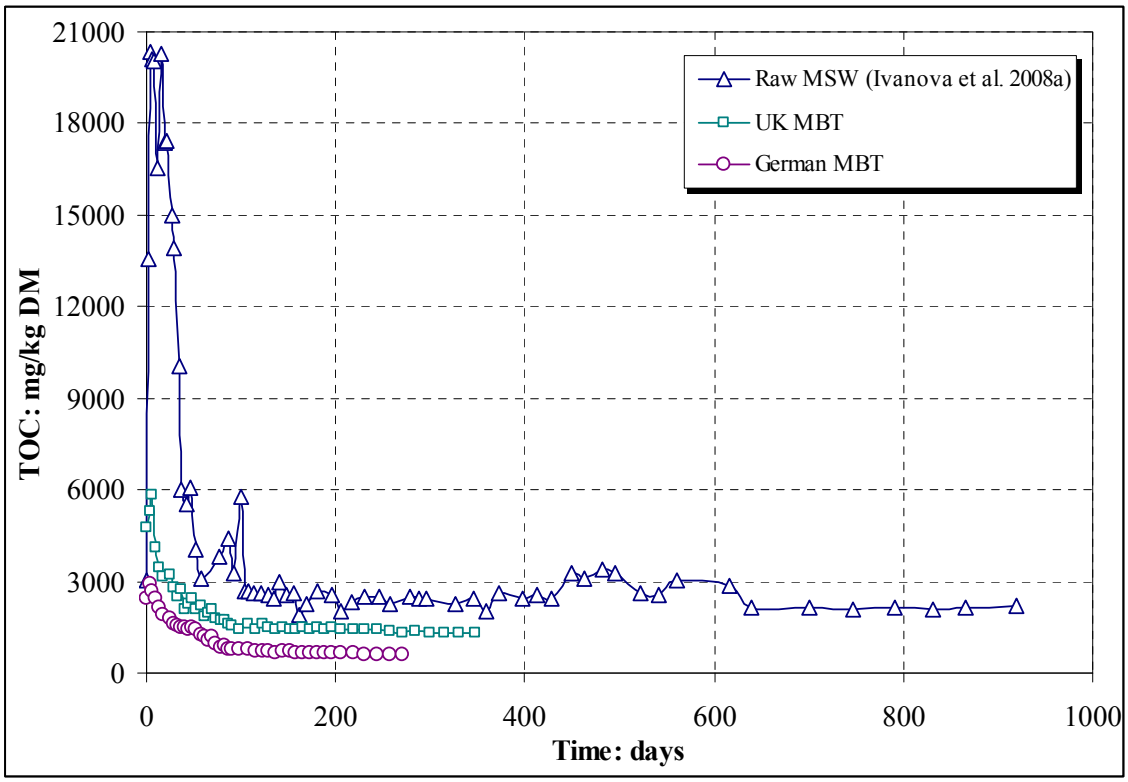


Figure 4.16: Leachate TOC load in raw MSW and pre-treated wastes

4.1.2.5 Ammoniacal and total nitrogen

Ammoniacal nitrogen (NH_4-N) is one of the pollutants found in leachate that causes great concern. If released from landfill, it would pose potentially serious threats to the surrounding soil and water such as eutrophication, toxicity to aquatic life and contamination of groundwater. Figure 4.17 presents the ammoniacal and total nitrogen concentrations in the leachate from CAR2 for the UK and German MBT wastes. Total nitrogen in theory is the sum of nitrite, nitrate, ammoniacal and organic nitrogen. Since the nitrite and nitrate nitrogen are expected to be low under anaerobic conditions, and negative redox potential, the total nitrogen composed of mainly ammoniacal and organic nitrogen in the leachate. Therefore, the total nitrogen (TN) trend was to a large extent identical to that of the ammoniacal nitrogen trend.

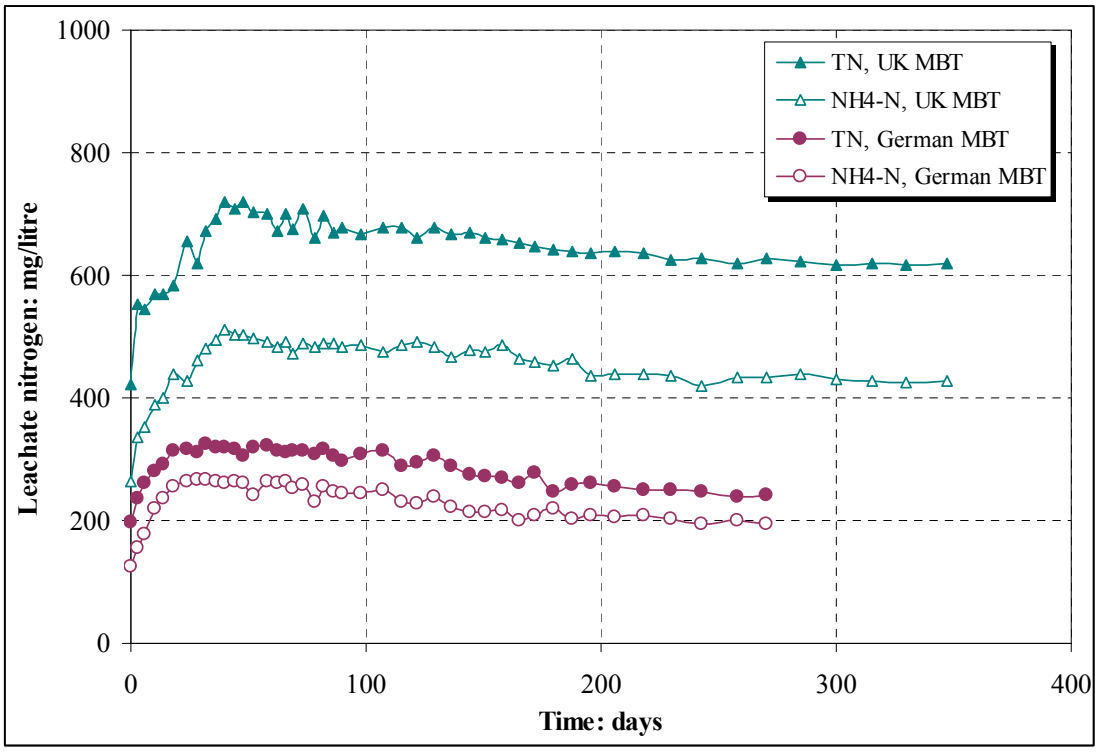


Figure 4.17: Leachate nitrogen in CAR2 for the UK and German MBT wastes

The concentration of TN and NH_4-N increased in the leachate with time, in contrast to the decrease of organic carbon i.e. TOC. The initial sharp increase in TN and NH_4-N may be explained by the direct leaching of ammonia from the waste, and the microbial degradation of nitrogenous organics like protein and amino acids (Berge et al., 2005; Jokela and Rintala, 2003). Most of the TN was found to be in the form of ammoniacal nitrogen. After an initial increase, TN and NH_4-N remained stable for some period as reported in other studies of the landfills with leachate recirculation (Price et al., 2003; Onay and Pohland 1998). Ammoniacal nitrogen concentration then decreased very slowly to a stable concentration of about 425 mg/litre and 195 mg/litre for the UK and German MBT wastes respectively. A similar decrease of ammonia has been observed but not fully explained in bioreactor landfill experiments (Sponza & Agdag, 2004; Bilgili et al., 2007). The only possible explanations for a slight decrease in NH_4-N are sorption in to the waste mass; anaerobic ammonium oxidation (anammox) for conversion to nitrogen gas (Jun et al., 2009; Valencia et al., 2009; Berge et al., 2005); microbial uptake for the growth of new cells; and precipitation of nitrogen as struvite (Kabdasli et al., 2000). This is consistent with a previous report on anaerobic reactors (Zhong et al. 2009).

The total nitrogen load in the leachate lies within the range previously reported by Horing et al. (1999), but is in excess of the values observed by Leikam and Stegmann (1999) and Bayard et al. (2008). The results indicated that the leaching potential of nitrogen from the German MBT waste was less than that from the UK MBT waste. This is probably due to the lower TN content of the German MBT waste. The TN concentration of the leachate in

CAR1 was more stable, remaining in the range 200-300 mg/litre throughout the experiment (Figure 4.18).

Figure 4.19 compares the trend of ammoniacal nitrogen load in the leachate obtained in the current research with that obtained by Ivanova et al. (2008*a*) for raw MSW. Unfortunately, information on the TN content in the leachate for raw MSW was not available. Ammoniacal nitrogen was present in a high concentration in the leachate from raw MSW. Nonetheless, direct comparison of leachate quality (i.e. NH_4-N concentration in mg/litre) is inappropriate as already discussed. Contrasting results were found for raw MSW as the NH_4-N concentration remained nearly constant until 600 days had elapsed: there then followed a sharp increase, which was attributed to the death/lysis of the bacterial biomass.

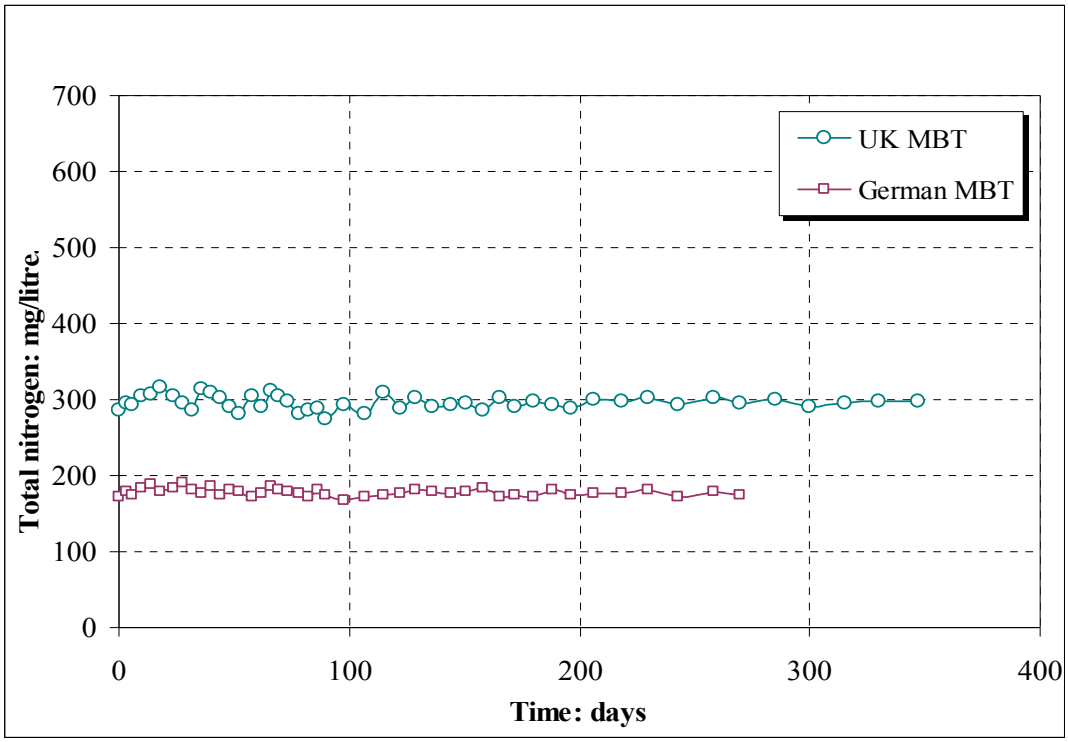


Figure 4.18: Total nitrogen in the leachate from CAR1 for the UK and German MBT wastes

A significant benefit of the waste pretreatment is evident from the levels of ammoniacal nitrogen, which are substantially lower for the pre-treated wastes than for raw MSW. Therefore, pretreatment will contribute to the significant reduction of this aspect of a landfill's polluting potential.

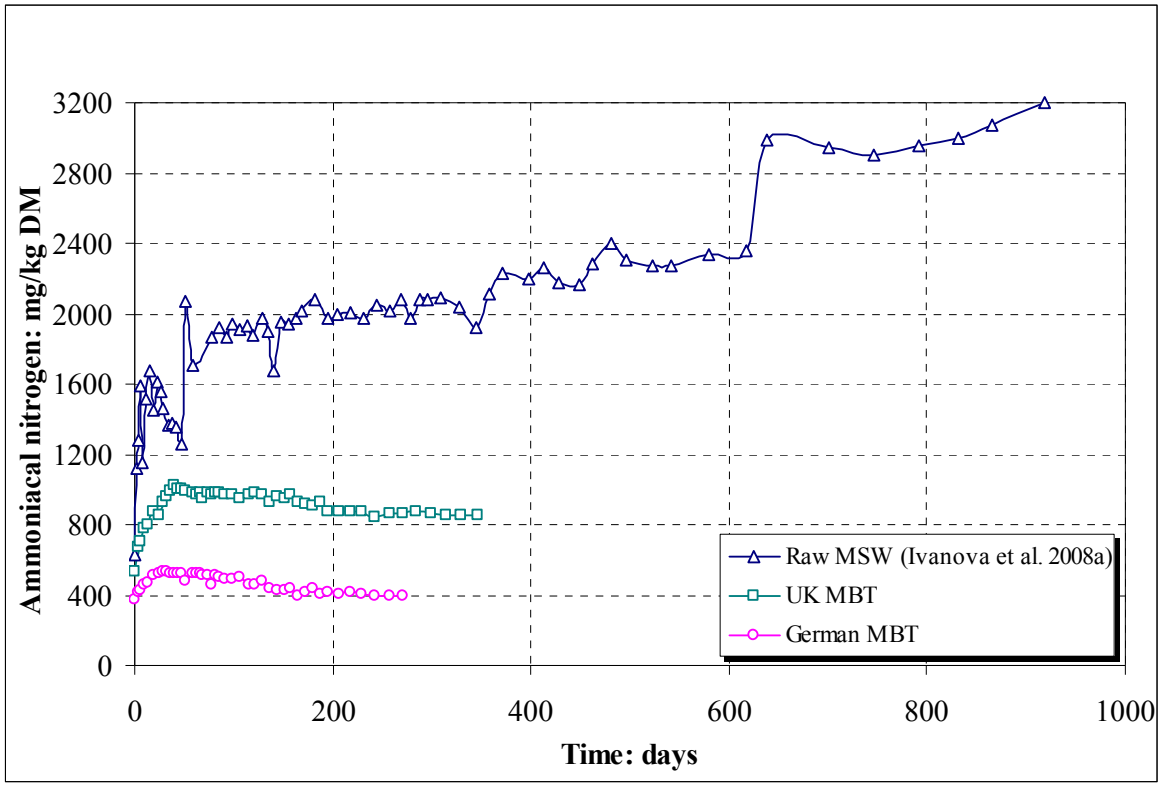


Figure 4.19: Leachate NH_4-N load in raw MSW and pre-treated wastes

4.1.2.6 Heavy metals

Leachate samples were analysed for six heavy metals: Zinc (Zn), Nickel (Ni), Copper (Cu), Chromium (Cr), Cadmium (Cd) and Lead (Pb). The changes in concentrations of these heavy metals in the leachate from CAR2 for the UK MBT waste are plotted in Figs. 4.20 & 4.21, and for the German MBT waste in Figs. 4.22 & 4.23. The curves obtained for the two MBT residues were quite similar in shape, even if the actual values of heavy metals released were different. The dissolved heavy metals concentrations in the leachate samples (which had previously been filtered through Whatman GF/C filters) were observed to decrease gradually during the experiment. Concentrations were generally low except for zinc and nickel.

After the onset of methanogenesis, metal concentrations tend to decrease owing to the establishment of a highly reducing environment (confirmed by low redox potential values). A significant decrease in all metal concentrations, except for chromium and cadmium occurred. Immobilisation of heavy metals from the soluble phase was completed in about 50 days, and concentrations then remained relatively constant till the end of the experiment. Possible metal depletion or immobilisation processes are the formation of insoluble precipitates, principally sulphides or carbonates, and sorption to refuse and suspended solids. Dissolved chromium would be expected to form insoluble hydroxides at the pH and ORP conditions of CAR2. Other factors influencing metal concentrations in the leachate include solubility, complex formation, surface adsorption and colloidal content (Christensen et al., 2001). Low concentrations of heavy metals in the leachate

primarily reflect the low amount of metals in the waste and their low solubility.

These data suggest that metal concentrations will not be problematic for the environment as the final concentrations were found to be less than or comparable with the European Drinking Water Standards (Council Directive 98/83/EC). Heavy metal concentrations were low in the leachate from the German MBT waste compared with that from the UK MBT waste. The low concentrations of heavy metals are in agreement with the results from other studies (van Praagh et al., 2009, Robinson et al., 2005).

The average metal concentrations obtained for raw MSW by Ivanova (2007) were 1.2, 0.45, 0.19, 0.15 and 0.003 mg/litre for Zn, Cu, Ni, Pb and Cd respectively. It could be inferred that leaching of heavy metals from raw MSW is more than that of pre-treated wastes. It seems that the pretreatment of MSW may have either reduced the metal content in the waste or make the metals less mobile which is a positive effect.

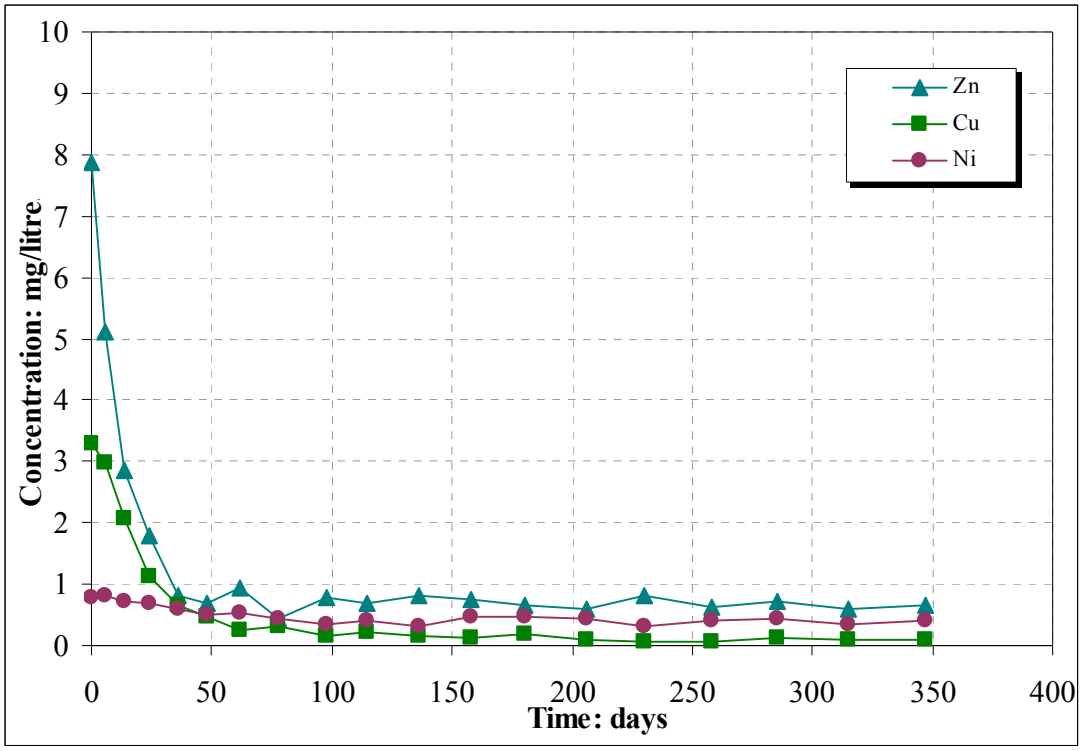


Figure 4.20: Zn, Cu and Ni concentration in leachate of the UK MBT waste in CAR2

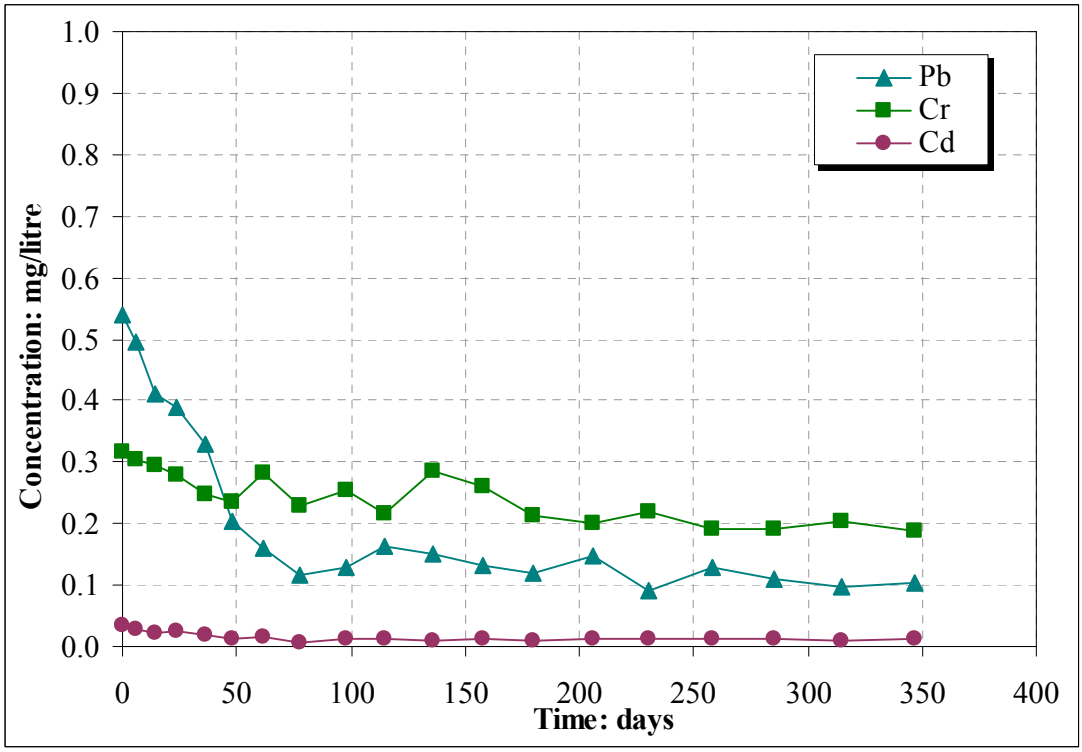


Figure 4.21: Pb, Cr and Cd concentration in leachate of the UK MBT waste in CAR2

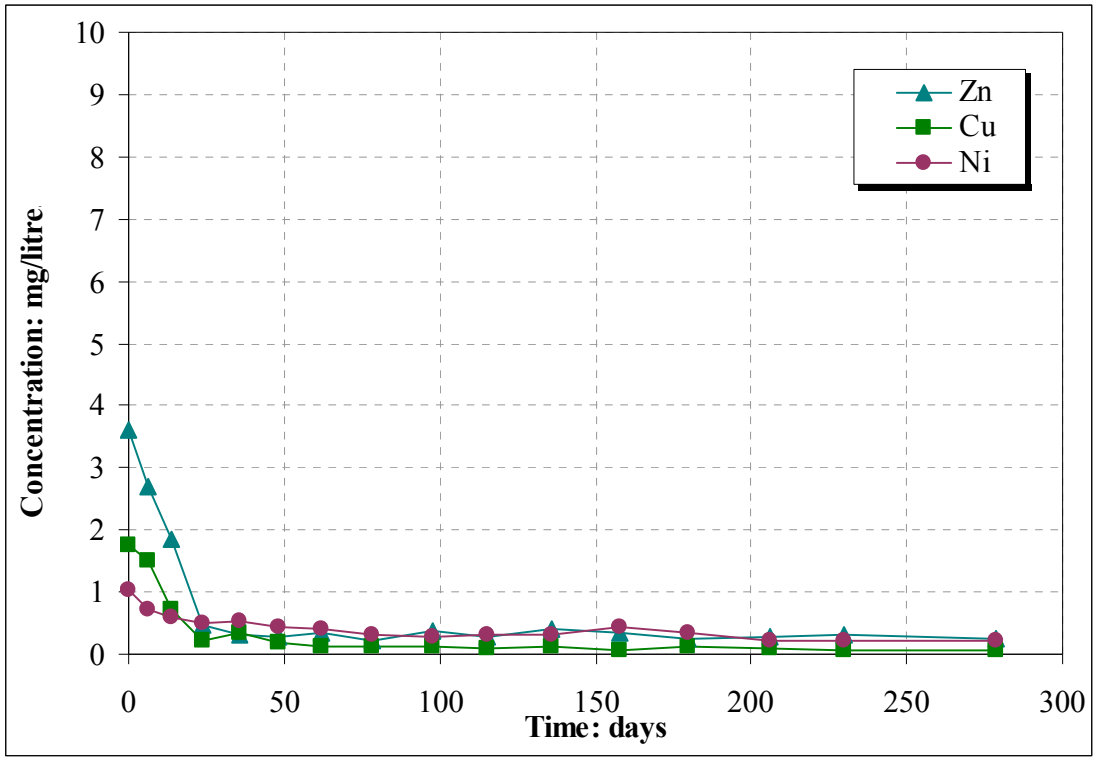


Figure 4.22: Zn, Cu and Ni concentration in leachate of the German MBT waste in CAR2

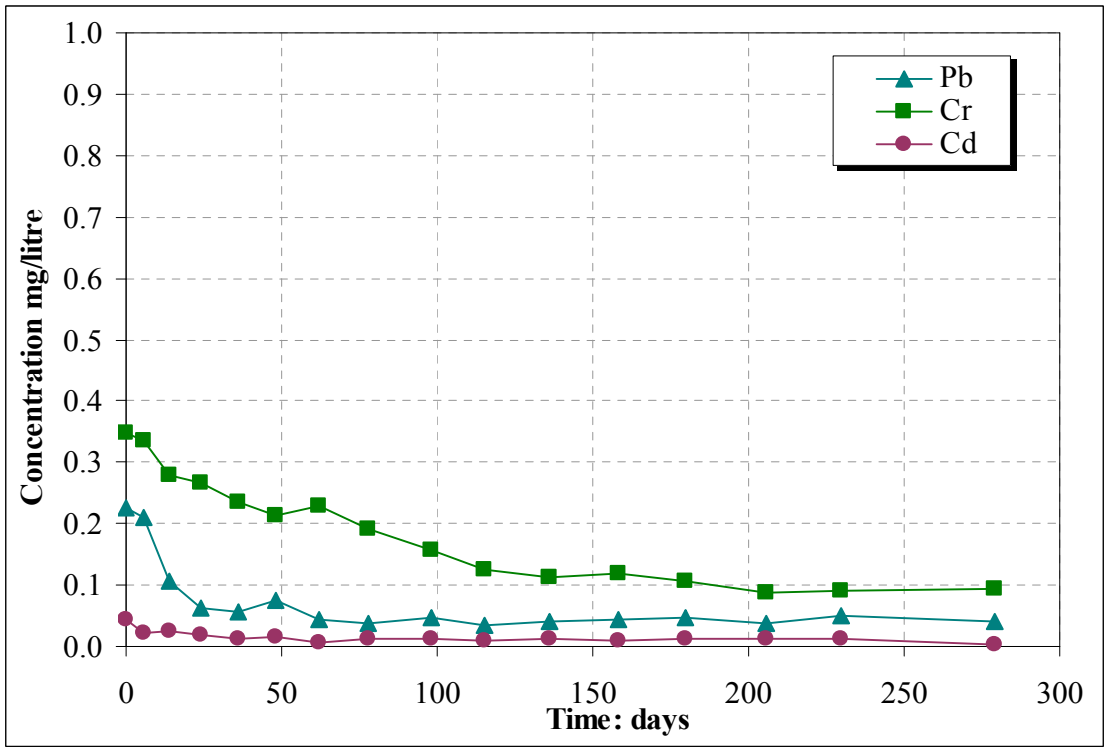


Figure 4.23: Pb, Cr and Cd concentration in leachate of the German MBT waste in CAR2

4.1.2.7 Chloride

Chloride is one of the major inorganic anions present in the leachate. It is a non degradable conservative and inert parameter, and would only be removed from landfills via washout. Chloride ion concentration is commonly used to assess leachate dilution (Bilgili et al., 2007), with an increase in chloride concentration suggesting that the leachate is becoming more concentrated. Kjeldsen (2002) reported no observable difference in chloride concentration between acidogenic and methanogenic phase leachates due to the minimal effects of sorption, complexation and precipitation. Figure 4.24 shows the chloride concentrations in the leachate of the CAR2 for the UK and German MBT wastes. Chloride ion concentration increased at the start of the experiment and stayed relatively constant with perhaps a very slow decline. A few slight jumps and fluctuations in the chloride concentration data of the UK MBT waste possibly resulted from the interference of bromide ions added as a part of a tracer study.

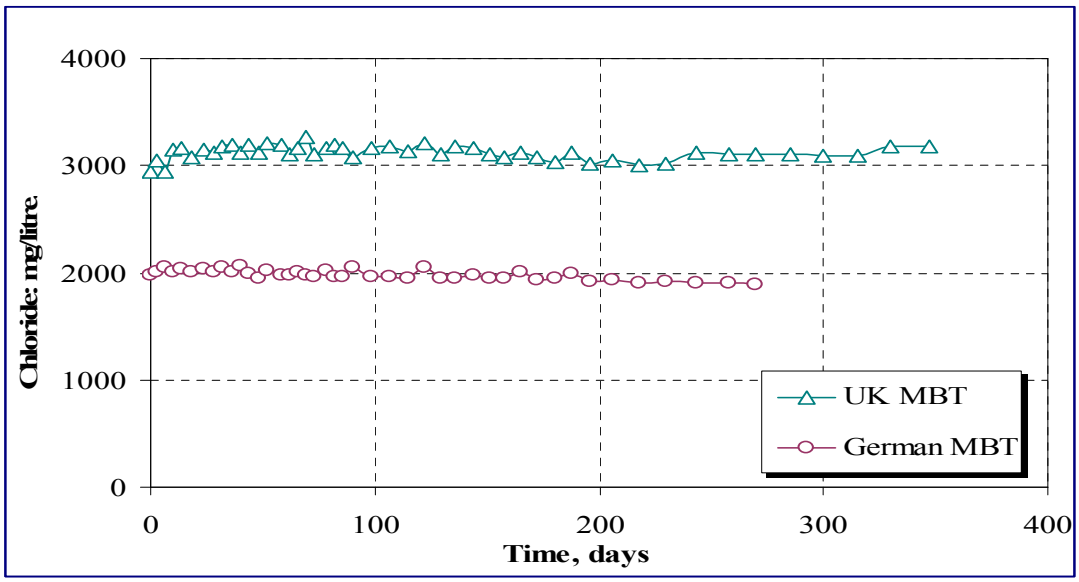


Figure 4.24: Chloride concentration in the leachate from CAR2 for the UK and German MBT wastes

Chloride concentrations are within the range reported by Robinson et al. (2005) and Kjeldsen et al. (2002). The chloride load (~ 6 g/kg DM for UK MBT and 4 g/kg DM for German MBT) is at the higher end of the range defined by Horing et al. (1999) but still lower than the 10 g/kg DM observed by Boni et al. (2003) for MBT waste.

4.1.2.8 Calcium and magnesium

The concentrations of calcium (Ca) and magnesium (Mg) in the leachate were studied to facilitate the carbon mass balance calculations. Ca and Mg in the leachate are sourced from the degradation of organic matter as well as dissolving inorganic waste. Their concentration is controlled by pH and carbonate ions. These ions are known to be involved in the precipitation of calcite ($CaCO_3$) and dolomite ($Ca\ Mg\ (CO_3)_2$) (Christensen et al. 1994). Figures 4.25 shows nearly the same trend for calcium and magnesium concentrations in leachate from CAR2 for the UK and German MBT wastes.

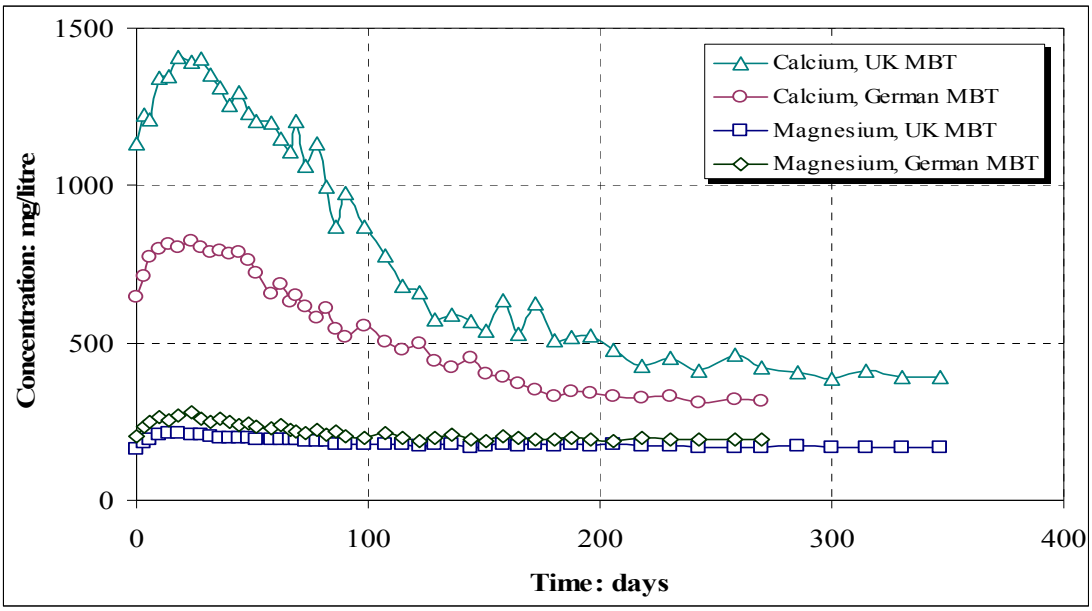


Figure 4.25: Calcium and magnesium concentrations in CAR2

The initial high calcium and magnesium concentrations and subsequent decrease could indicate that initially the ions were leached from the solid refuse into leachate, followed by the precipitation as carbonates. The calcium and magnesium concentrations in the leachate are in the range reported by Robinson et al. (2005) and Kjeldsen et al. (2002).

4.1.3 Leachate volumes in pond and waste

The volume of leachate in the upper pond in CAR2 was 39.8 litres for the UK MBT waste and 40.3 litres for the German MBT waste. The remaining 40.2 litres and 39.7 litres of leachate was held within the waste and the gravel layers. It should be borne in mind that the CAR experiments were therefore carried out with 100% excess leachate, doubling the liquid to solid ratio compared with a field condition in which only the pores in the waste were filled with leachate. Assuming that the same amount of contaminant was washed out of the waste, the effect of this would be to double contaminant concentrations in the field.

4.1.4 Solids composition

At the end of each experiment, a degraded waste sample was obtained from CAR2 by scraping off the top 5 cm layer of waste. This was done to retain the remaining waste in the CAR for further research on waste structure and flow properties (by others). The degraded waste sample retrieved from CAR2 was dried and washed with distilled water to remove any attached biomass. The washed sample was oven dried at 70^{o}C and analysed for LOI, TC, TN, cellulose, hemicellulose and lignin contents. The data are summarized in Tables 4.1 and 4.2 for the UK and German MBT wastes respectively.

4.2 Small scale BMP reactors

BMP tests were carried out to characterise the anaerobic biodegradation potential of MBT wastes. This characterisation included measurement of biogas yield, solids composition (LOI, TC, C, H and L contents), and assessment of leachate characteristics at various stages of biodegradation process over periods of 347 and 279 days for the UK and German MBT wastes respectively. The experiment was carried out as outlined in Section 3.5.2. The BMP reactors were terminated sequentially during the biodegradation process and waste, leachate and gas were sampled and analysed. Periodic sampling from the reactors allowed the leachate and solid waste compositional changes to be tracked. Solids and associated leachate were collected separately from each of the reactors by filtration through GF/C filters (Whatman). Leachate samples were analysed for pH, VFA, TOC, DOC, IC, ammoniacal nitrogen, TN, chloride, calcium and magnesium ions. The solid waste samples were analysed for LOI, TC, TN, cellulose (C), hemicellulose (H), and lignin (L) contents to characterize the change in solids composition.

4.2.1 Biogas

Cumulative gas production measured from the BMP reactors is shown in Figs. 4.26 & 4.27 for the UK and German MBT wastes respectively. The net biogas yield attributable to 140 g dried waste sample in each reactor was determined by subtracting the measured biogas yield of the control bottles from the total biogas produced by each reactor. The volume of biogas collected was standardised to dry gas at STP. The gas production curves for

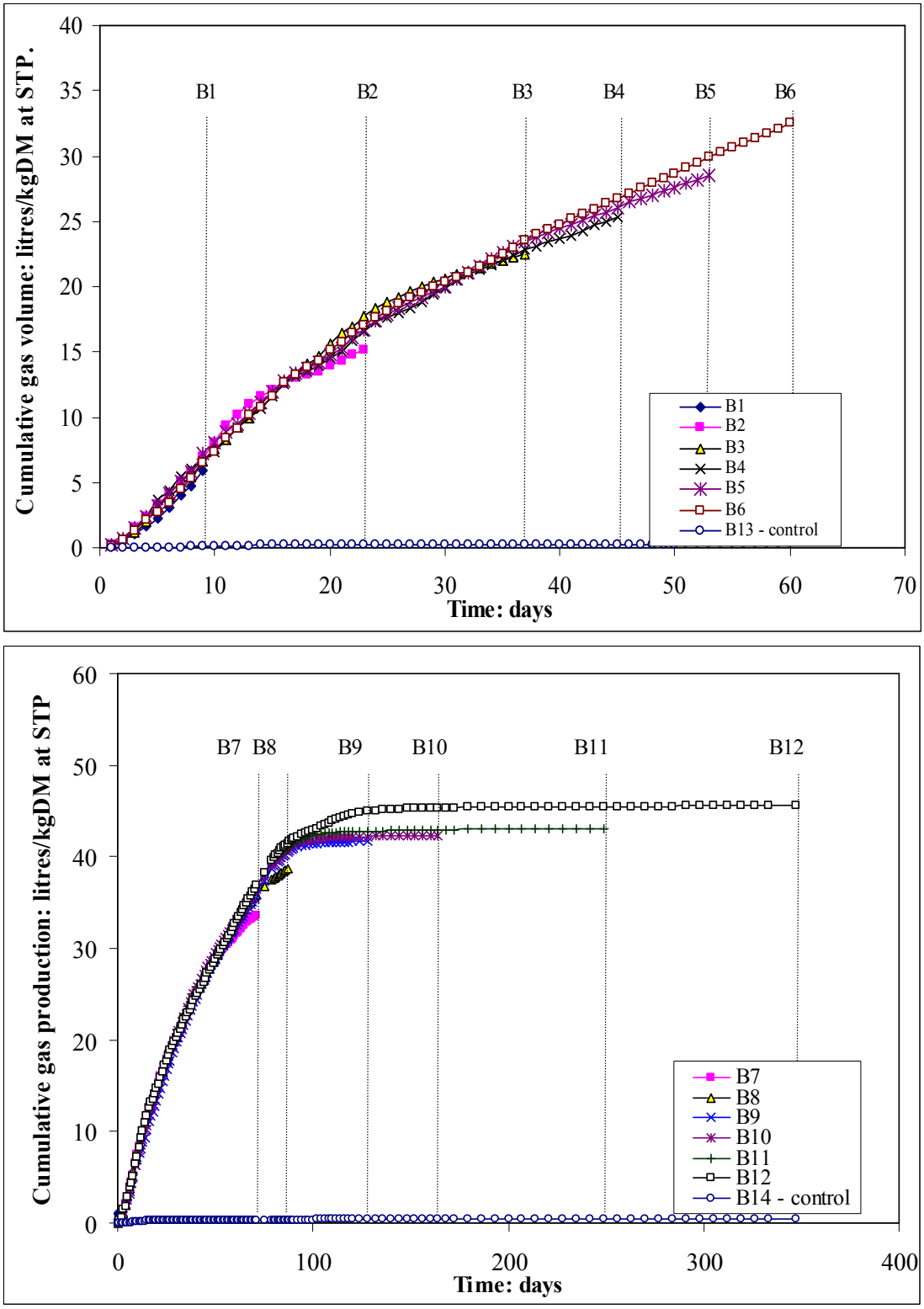


Figure 4.26: Cumulative gas production in BMP reactors for the UK MBT waste

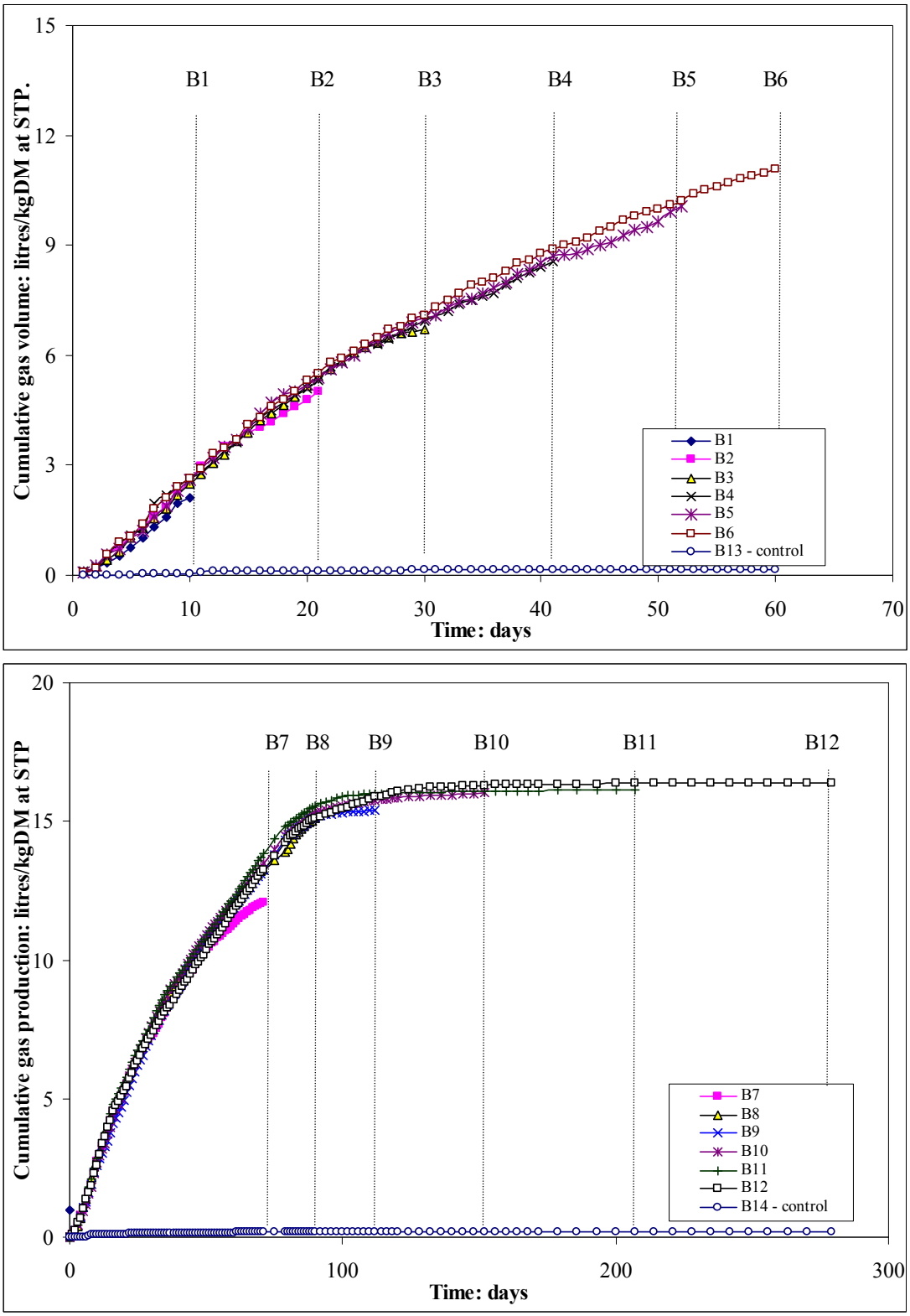


Figure 4.27: Cumulative gas production in BMP reactors for the German MBT waste

BMP reactors (B1- B12) were almost identical confirming homogeneity of the waste and the test conditions. Biogas production for the UK and German MBT wastes increased in a similar trend at different rates during the first 100 days. However, the UK MBT waste produced considerably more biogas, reaching approximately 45.54 litre/kg DM after 347 days. In contrast, biogas production from the German MBT waste was 16.39 litre/kg DM after 279 days. The methane content of the biogas ranged between 58 to 62% as shown in the Figs. 4.28 & 4.29. Details of the cumulative biogas produced for the BMP reactors are given in Tables 4.1 and 4.2 for the UK and German MBT wastes respectively.

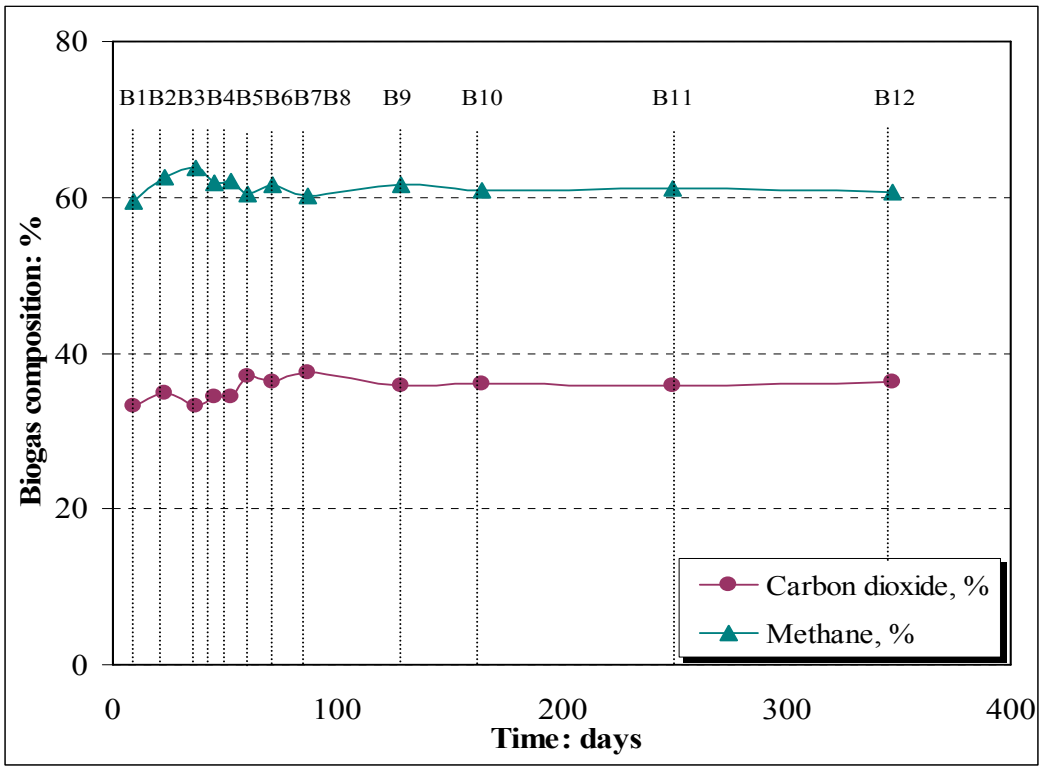


Figure 4.28: Biogas composition in the BMP reactors for the UK MBT waste

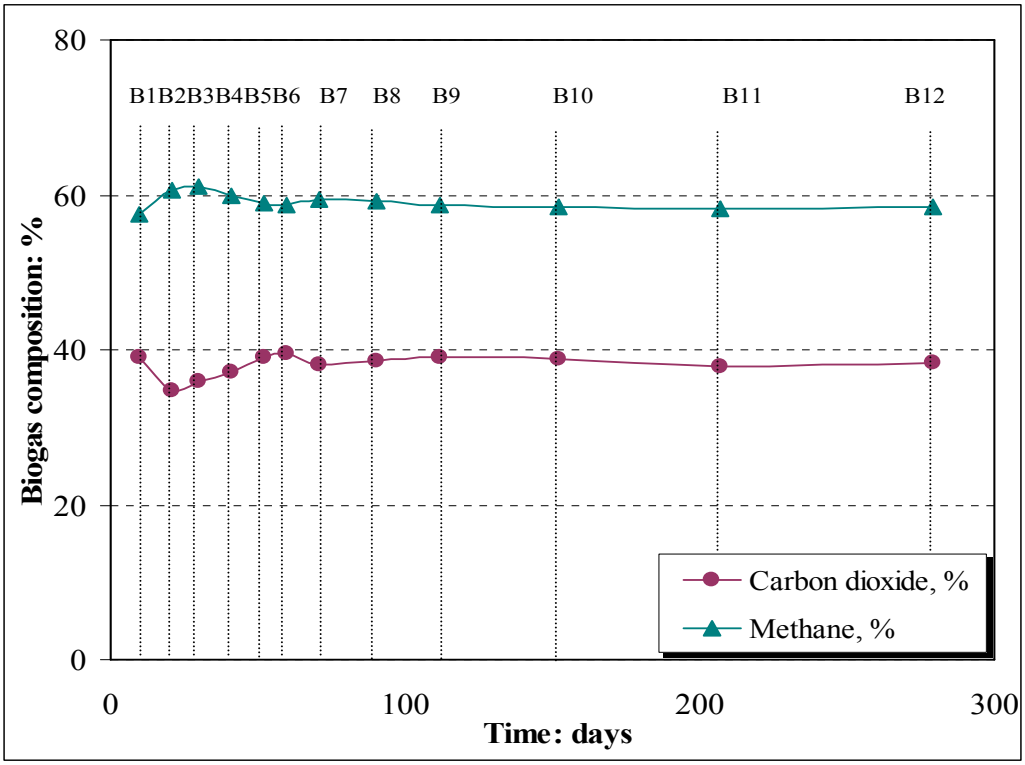


Figure 4.29: Biogas composition in the BMP reactors for the German MBT waste

4.2.2 Leachate quality

The characteristics of the leachate samples for the UK and German MBT wastes taken from the BMP reactors are discussed below.

The pH values were increased slightly at the start, possibly due to a decrease in VFA concentration, and then remained stable in the range 7.5 to 7.7 (Figs 4.30 & 4.31). The decrease in VFA concentration is consistent with an increase in pH and biogas production. The TOC content of the leachate samples taken from the reactors decreased gradually to about 465 mg/litre and 198 mg/litre by the end of the test for the UK and German MBT wastes respectively (Figs. 4.32 & 4.33). DOC showed a similar trend to the TOC indicating that carbon is mostly in dissolved form.

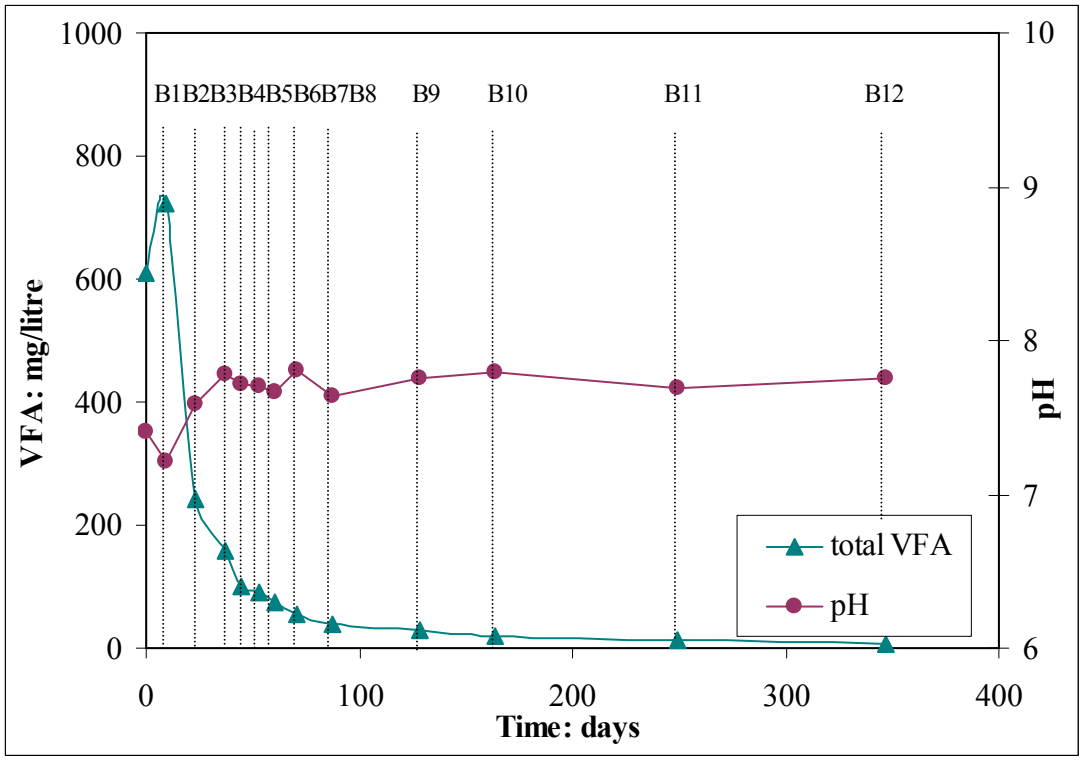


Figure 4.30: Leachate VFA and pH in BMP reactors for the UK MBT waste

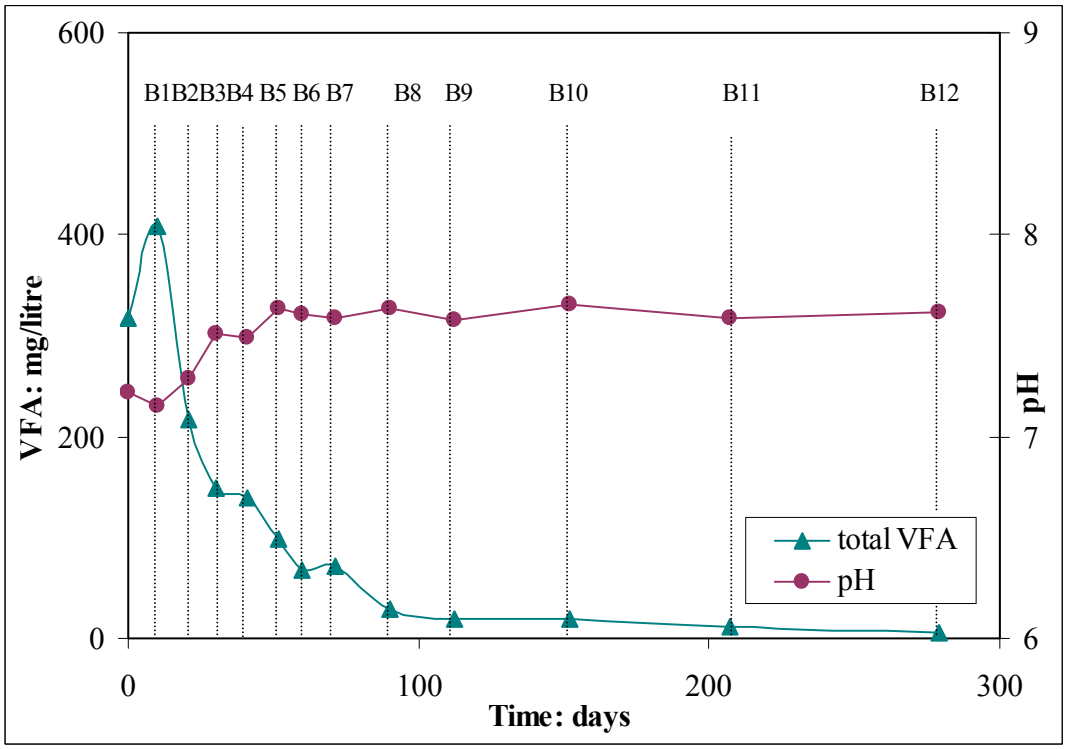


Figure 4.31: Leachate VFA and pH in BMP reactors for the German MBT waste

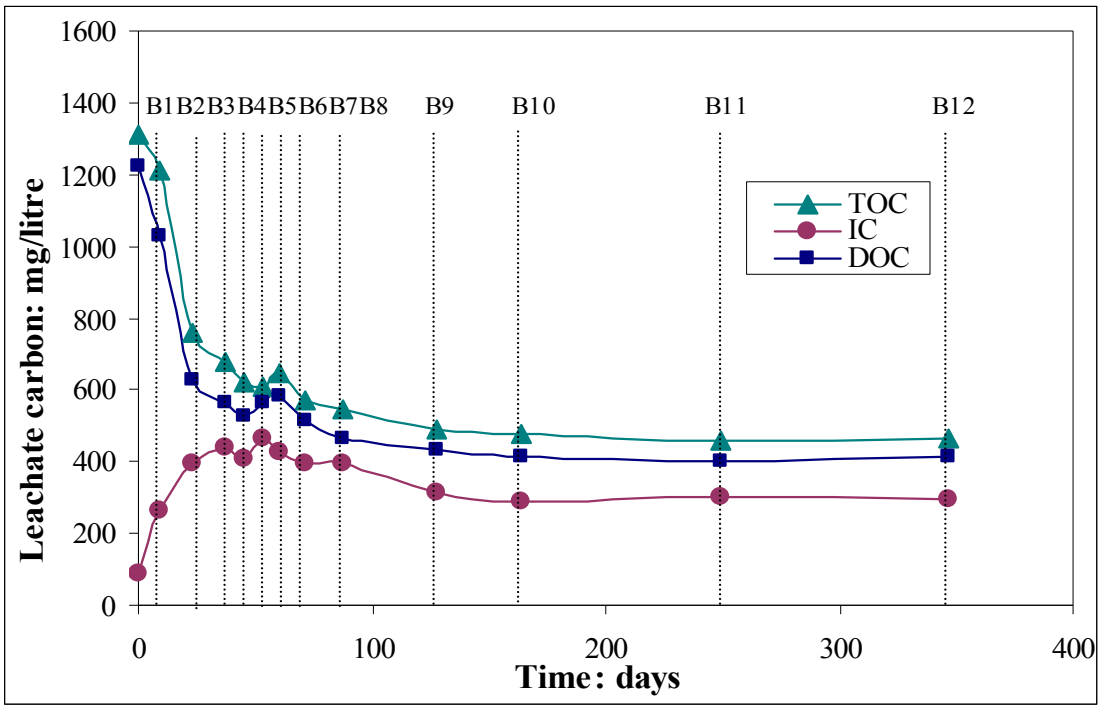


Figure 4.32: Leachate carbon in BMP reactors for the UK MBT waste

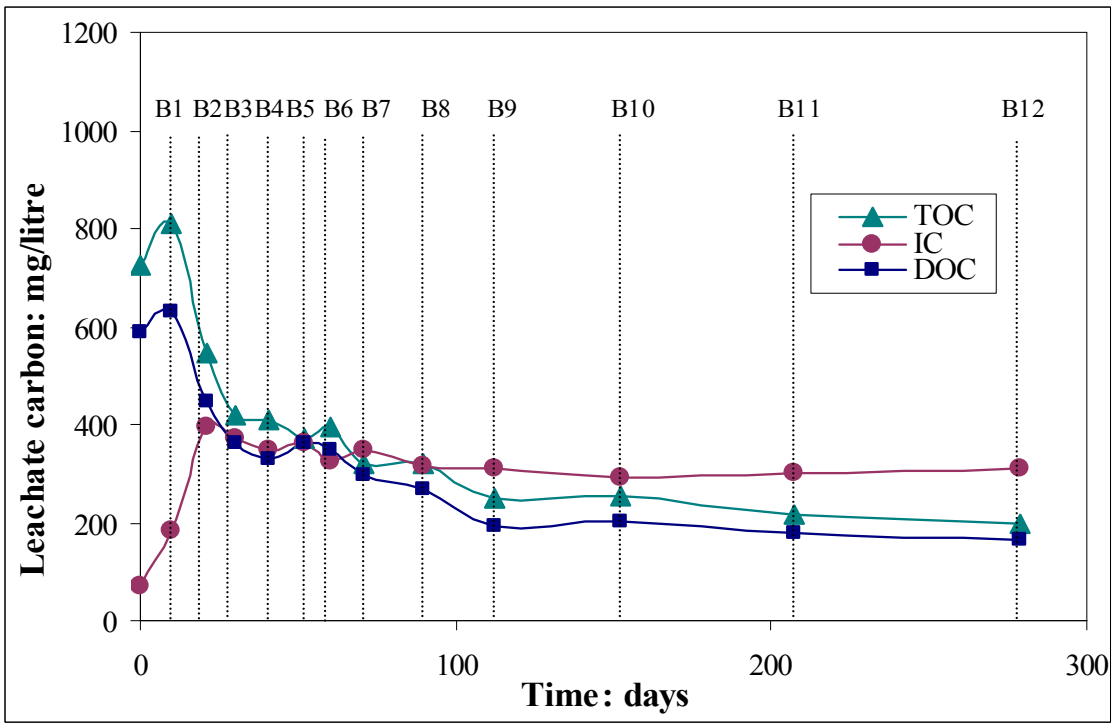


Figure 4.33: Leachate carbon in BMP reactors for the German MBT waste

The concentrations of ammoniacal and total nitrogen in the leachate sampled from the BMP reactors are presented in Figs. 4.34 & 4.35 for the UK and German MBT wastes respectively. The concentrations increased primarily due to the dissolution and breakdown of organic matter from the solid to the liquid phase, and remained stable thereafter. The trends are similar for the UK and German MBT wastes but the concentrations are different.

The calcium concentrations first increase to a peak and then follow a slight decreasing trend, possibly due to their precipitation as carbonates (Fig. 4.36). The magnesium concentrations were much more steady.

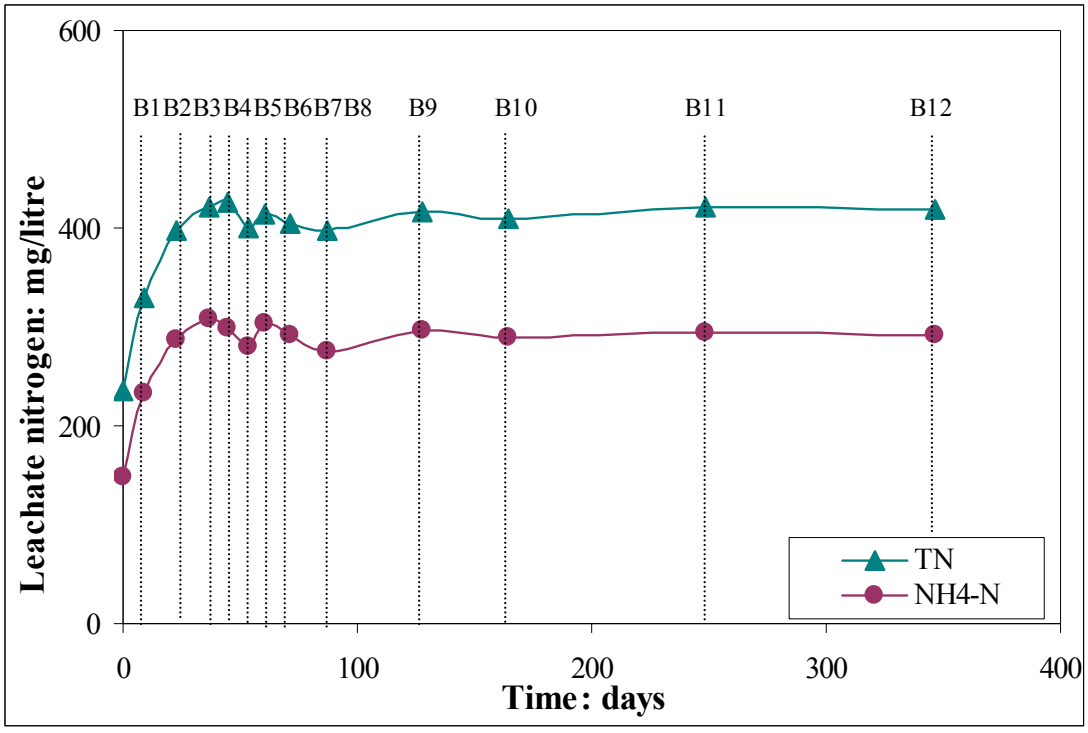


Figure 4.34: Leachate nitrogen in BMP reactors for the UK MBT waste

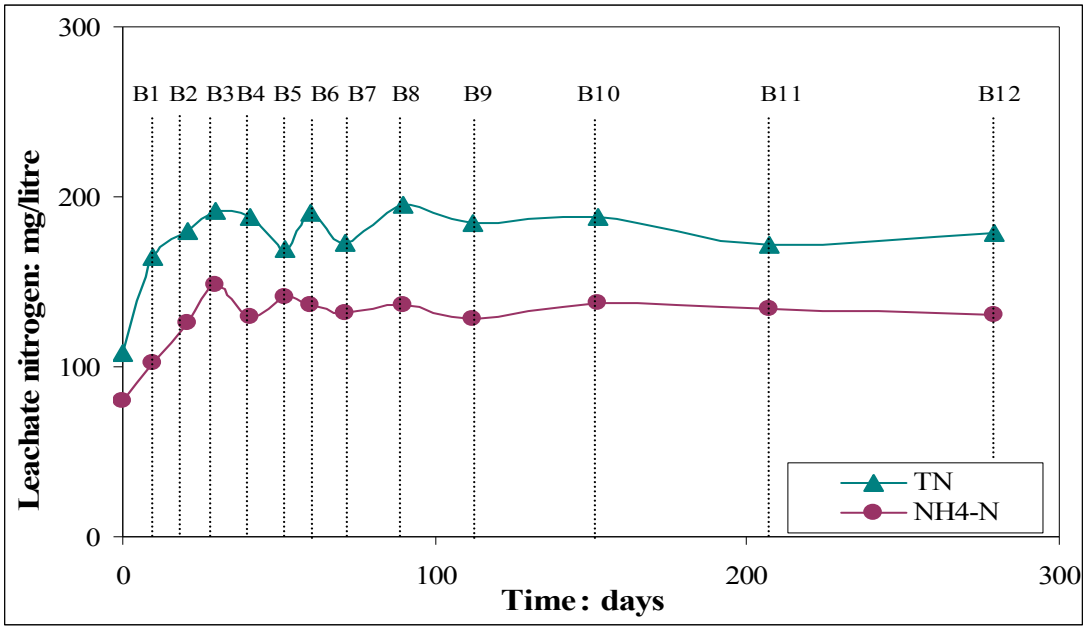


Figure 4.35: Leachate nitrogen in BMP reactors for the German MBT waste

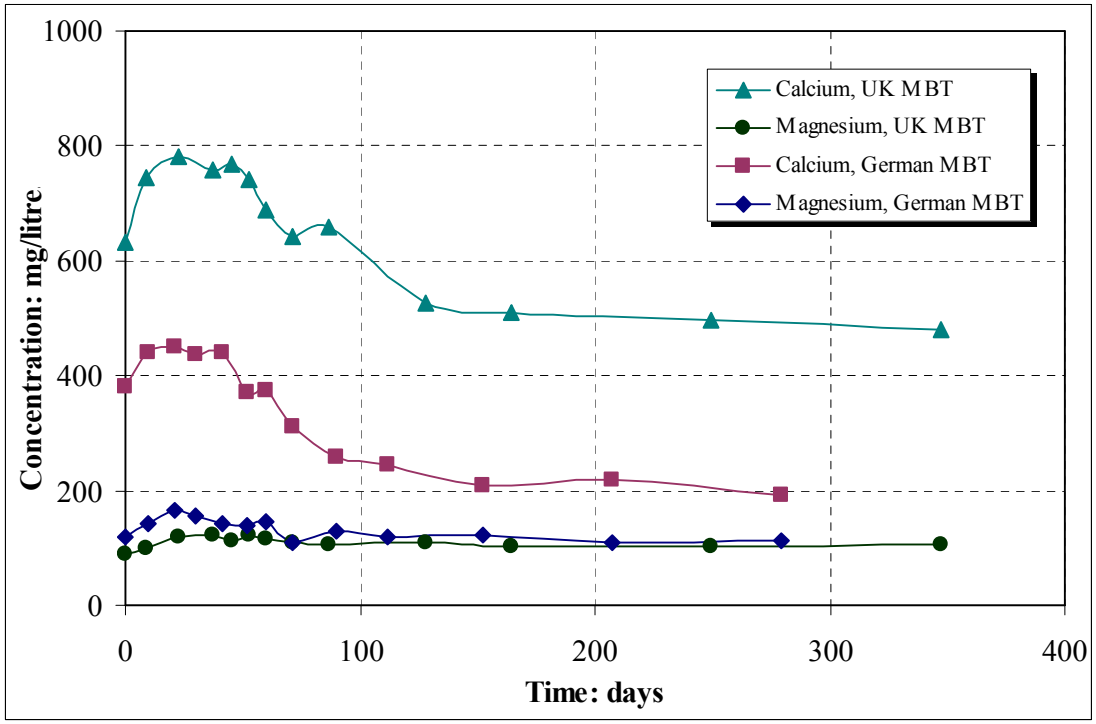


Figure 4.36: Calcium and magnesium concentrations in the BMP reactors for the UK and German MBT wastes

It is interesting to note that the concentration (mg/litre) of TOC, NH_4-N etc. in the leachate of BMP reactors is less as compared to leachate in CAR2, but the leachate loads (g/kg DM) are similar for both BMP reactors and CAR2. This implies that the comparison of leachate strength amongst different studies would be better in terms of the leachate load if L/S ratios are known.

4.2.3 Solids composition

The initial solids compositions of the UK and German MBT wastes e.g. LOI, TC, TN, cellulose, hemicellulose and lignin were the same for the BMP reactors and CAR2 (Table 3.3). Data on the chemical composition of the degraded waste samples retrieved from the BMP reactors at different stages of the biodegradation process and from CAR2 at the end of experiment are summarised in Tables 4.1 and 4.2 for the UK and German MBT wastes respectively. All the data for loss on ignition (LOI), neutral detergent fibre (NDF), acid detergent fibre (ADF), cellulose, hemicellulose, and lignin content of the waste are presented as a percentage of the waste dry mass remaining in each reactor. The degradability of the waste decomposed in BMP reactors was assessed by measuring the biogas potential, LOI, TC, cellulose, hemicellulose and lignin contents.

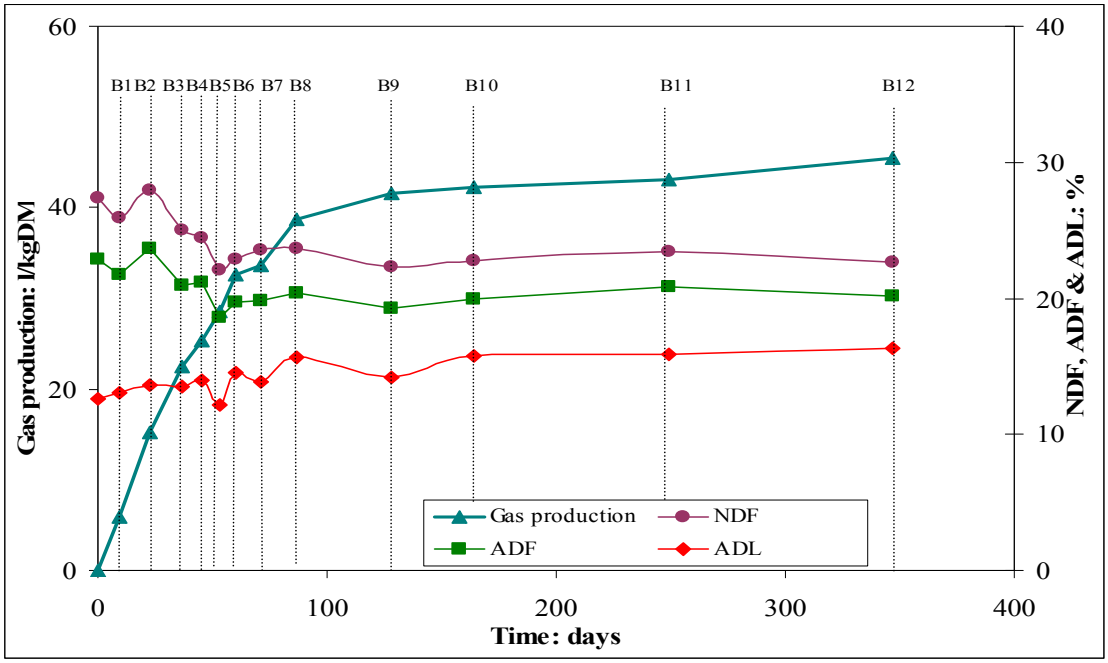


Figure 4.37: Change of NDF, ADF and ADL contents during gas production in BMP reactors for the UK MBT waste

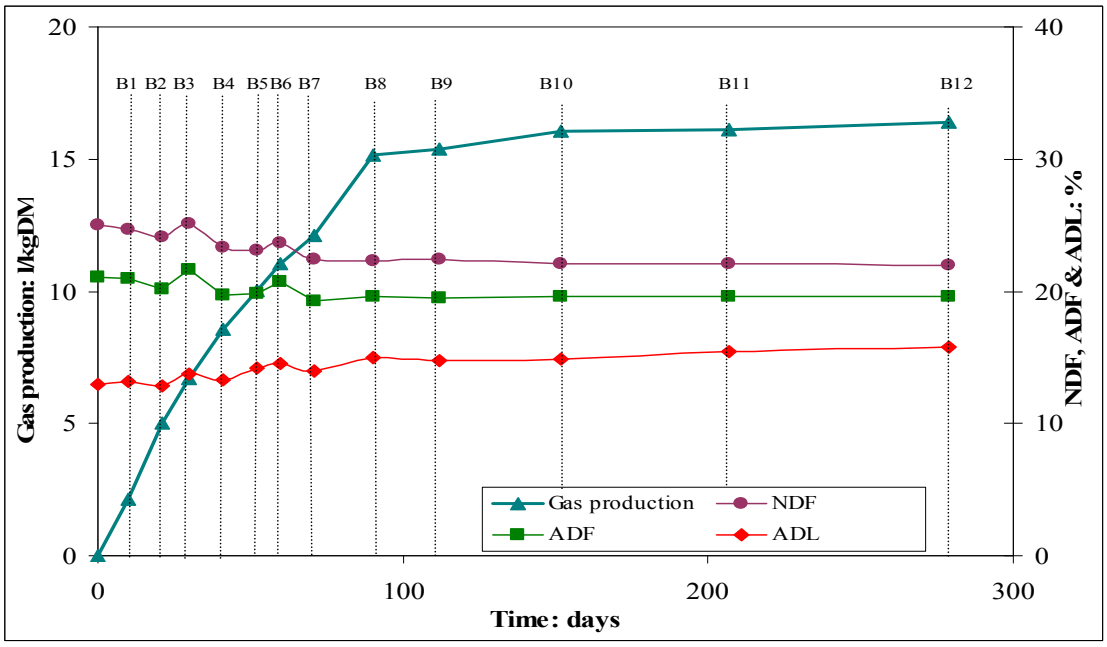


Figure 4.38 Change of NDF, ADF and ADL contents during gas production in BMP reactors for the German MBT waste

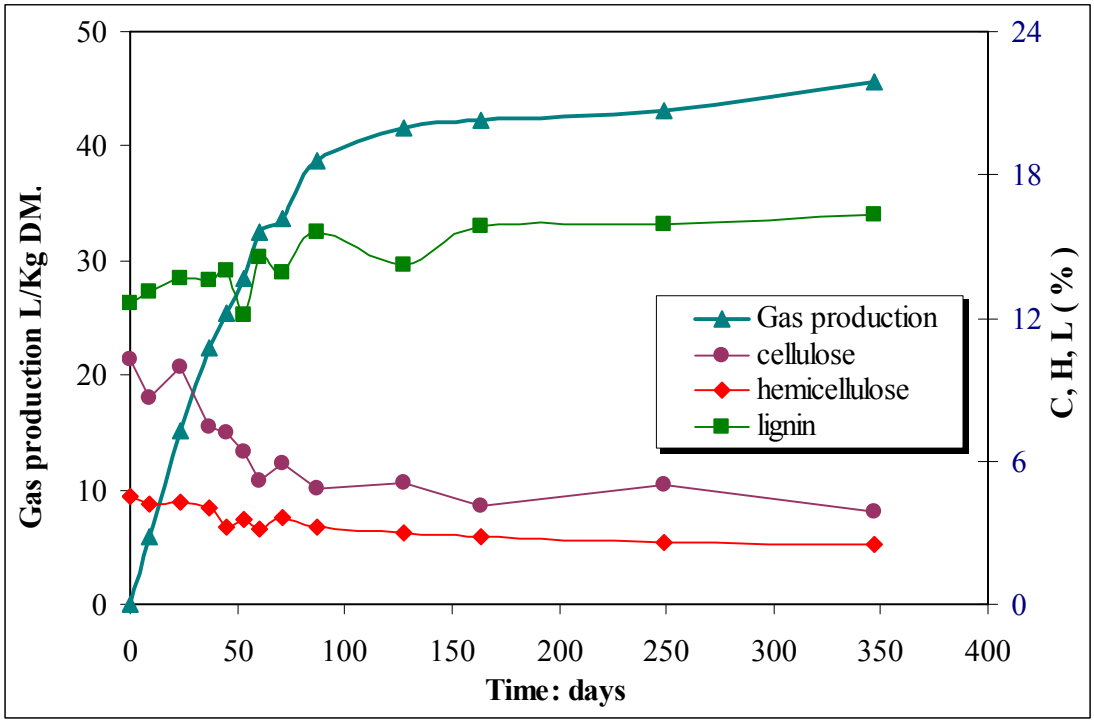


Figure 4.39: Change of cellulose, hemicellulose and lignin contents during gas production in BMP reactors for the UK MBT waste

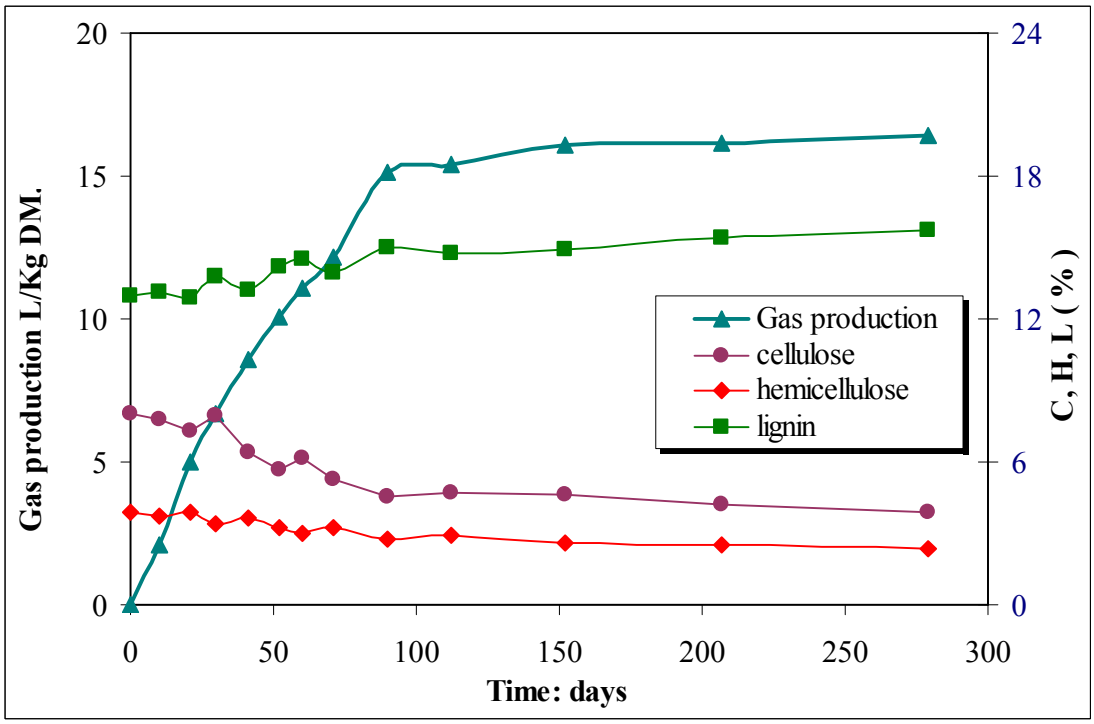


Figure 4.40: Change of cellulose, hemicellulose and lignin contents during gas production in BMP reactors for the German MBT waste

Table 4.1: Biogas production and fibre analysis data for the UK MBT waste in CAR2 and BMP reactors

	Initial waste	Waste sample												Final waste CAR2
		B1	B2	B3	B4	B5	B6	B7	B8	B9	B10	B11	B12	
Days into test	0	9	23	37	45	53	60	71	87	128	164	249	347	347
Dry mass, g (BMP reactors)	140.0	138.1	135.2	136.1	132.0	134.5	132.3	130.2	131.6	130.0	129.6	128.5	126.1	
Dry mass, kg (CAR2 reactor)	40.00													37.42
Cumulative biogas production , l/kg DM	0.00	5.93	15.18	22.43	25.36	28.50	32.55	33.59	38.65	41.64	42.28	43.07	45.54	49.46
NDF, %	27.41	25.95	27.89	24.96	24.42	22.14	22.87	23.51	23.69	22.26	22.80	23.49	22.68	23.31
ADF, %	22.87	21.76	23.61	20.95	21.16	18.55	19.69	19.84	20.44	19.28	19.96	20.88	20.14	21.25
ADL (Lignin), %	12.63	13.11	13.69	13.54	13.97	12.13	14.54	13.91	15.62	14.19	15.81	15.90	16.29	17.14
LOI, %	42.91	41.84	39.65	40.22	36.45	34.90	39.15	32.79	32.64	30.97	31.55	30.71	30.64	29.38
Cellulose, %	10.24	8.65	9.92	7.41	7.19	6.42	5.15	5.93	4.82	5.09	4.15	4.98	3.85	4.11
Hemicellulose, %	4.54	4.19	4.28	4.01	3.26	3.59	3.18	3.67	3.25	2.98	2.84	2.61	2.54	2.06
(C+H)/L ratio	1.17	0.98	1.04	0.84	0.75	0.82	0.57	0.69	0.52	0.57	0.44	0.48	0.39	0.36
TC, %	22.68	22.19	21.08	21.91	21.02	20.68	21.14	20.30	20.57	20.04	20.11	19.94	19.27	19.51
TN, %	1.81	1.72	1.70	1.74	1.69	1.72	1.66	1.62	1.65	1.64	1.61	1.63	1.61	1.59

Table 4.2: Biogas production and fibre analysis data for the German MBT waste in CAR2 and BMP reactors

	Initial waste	Waste sample												Final waste CAR2
		B1	B2	B3	B4	B5	B6	B7	B8	B9	B10	B11	B12	
Days into test	0	10	21	30	41	52	60	71	90	112	152	207	279	279
Dry mass, g (BMP reactors)	140.0	138.8	136.4	136.7	135.1	134.9	132.4	131.2	133.0	130.7	129.1	130.2	130.0	
Dry mass, kg (CAR2 reactor)	40.00													38.91
Cumulative biogas production , l/kg DM	0.00	2.12	5.01	6.70	8.55	10.05	11.07	12.13	15.15	15.39	16.05	16.13	16.39	17.74
NDF, %	24.88	24.65	24.11	25.12	23.29	23.08	23.70	22.46	22.33	22.4	22.14	22.10	22.02	21.93
ADF, %	20.97	30.93	20.21	21.68	19.67	19.87	20.68	19.22	19.58	19.48	19.55	19.62	19.66	19.72
ADL (Lignin), %	13.01	13.15	12.89	13.77	13.25	14.19	14.55	13.98	15.03	14.76	14.92	15.44	15.75	15.48
LOI, %	34.84	33.92	32.21	30.69	31.52	29.69	30.55	28.74	29.52	27.40	26.31	26.98	26.29	26.93
Cellulose, %	7.96	7.78	7.32	7.91	6.42	5.68	6.13	5.24	4.55	4.72	4.63	4.18	3.91	4.24
Hemicellulose, %	3.91	3.72	3.9	3.44	3.62	3.21	3.02	3.24	2.75	2.92	2.59	2.48	2.36	2.21
(C+H)/L ratio	0.91	0.87	0.87	0.82	0.76	0.63	0.63	0.6	0.48	0.52	0.48	0.43	0.40	0.42
TC, %	19.85	19.71	19.08	19.39	19.01	18.68	19.14	18.3	18.45	18.59	18.20	17.74	17.97	18.29
TN, % .	1.52	1.51	1.47	1.42	1.45	1.41	1.44	1.42	1.39	1.41	1.38	1.40	1.39	1.41

4.2.3.1 Theoretical methane calculation in CAR and BMP reactors

The stoichiometric relationships for the conversion of cellulose and hemicellulose to methane are presented in the following equations (Barlaz, 2006; Barlaz et al. 1990; Wagland et al., 2009):

Cellulose: $(C_6H_{10}O_5)_n + nH_2O \rightarrow 3nCH_4 + 3nCO_2$ Eq. 4.1

Hemicellulose: $(C_5H_8O_4)_n + nH_2O \rightarrow 2.5nCH_4 + 2.5nCO_2$ Eq. 4.2

Based on these equations, the theoretical methane production per gram of cellulose and hemicellulose degraded is 0.4148 litres and 0.4242 litres at STP, respectively (Barlaz et al., 1990; Barlaz, 2006). Using the cellulose and hemicellulose contents of the initial waste and final degraded waste in CAR2 and BMP reactors presented in Tables 4.1 and 4.2, the theoretical methane production per kilogram of initial dry mass of waste is calculated and compared with the measured methane production in Table 4.3.

Previous studies e.g. Angelidaki and Sanders (2004), Eleazer et al (1997), Wang et al. (1997) have shown that the practical yield obtained in a biogas reactor will always be lower and the degree of conversion upto 60% is more normal for the particulate organic matter. The discrepancy between the measured and the theoretical methane production can be partially explained by the fact that some carbon is utilized to synthesize bacterial mass, some is dissolved in the leachate and some is lost by biogas escape during sampling and injection of replaced leachate. The higher recovery of methane in UK waste could be due to the conversion of organics other than cellulose or hemicellulose to methane e.g. protein and fats that had not been accounted for in the original compositional analysis. UK waste probably has got more protein as reflected in the higher nitrogen content.

Table 4.3: Theoretical and measured methane production

Waste	Reactor	Theoretical methane production	Measured methane production
UK MBT	CAR2	37.32 litres/kg DM	29.68 litres/kg DM
	BMP	36.58 litres/kg DM	27.32 litres/kg DM
German MBT	CAR2	21.26 litres/kg DM	10.65 litres/kg DM
	BMP	20.32 litres/kg DM	9.84 litres/kg DM

Example calculation of theoretical methane for the UK MBT waste in CAR2:

Initial cellulose content = 10.24% of initial dry mass of waste

Final cellulose content = 4.11% of final dry mass of degraded waste

= 3.84% of initial dry mass of waste

Cellulose degraded (or cellulose loss)= 10.24% – 3.84%

= 6.4% of initial dry mass of waste

= 64 g/kg DM

Theoretical methane produced from degraded cellulose

$$= \frac{64\,\text{g cellulose}}{\text{kg DM}} \times \frac{0.4148\ \text{litres CH}_4}{\text{g cellulose}}$$

$$= \frac{26.54\ \text{litres CH}_4}{\text{kg DM}}$$

Initial hemicellulose content = 4.54% of initial dry mass of waste

Final hemicellulose content = 2 .06% of final dry mass of degraded waste

= 1.93% of initial dry mass of waste

Hemicellulose degraded (or loss in hemicellulose) = 4.54% - 1.93%

= 2.61% of initial dry mass of waste

=26.1 g/kg DM

Theoretical methane produced from degraded hemicellulose

$$= \frac{26.1\,\text{g hemi cellulose}}{\text{kg DM}} \times \frac{0.4242\,\text{litres CH}_4}{\text{g hemi cellulose}}$$

$$= \frac{10.82\ \text{litres CH}_4}{\text{kg DM}}$$

Total theoretical methane produced from degraded cellulose and hemicellulose

= 26.54 + 10.82 = 37.32 litres/kg DM

Measured methane production = 49.46 litres/kg DM x 0.6 = 29.68 litres/kg DM

4.2.3.2 *Chemical composition of solid waste samples*

The NDF and ADF values show a decreasing trend while ADL increased (becoming a greater proportion of the remaining waste) because lignin is a recalcitrant material (Figs. 4.37 & 4.38). This is consistent with the depletion of cellulose and hemicellulose as about 62% and 51% of cellulose and 45% and 40% of hemicellulose were degraded over the whole period of test for the UK and German MBT wastes. The cellulose content decreased from 10.2% to about 3.8% and from 7.9% to 3.9% for the UK and German MBT wastes respectively. The hemicellulose content decreased from 4.5% to about 2.5% and from 3.9% to 2.4% for the two wastes respectively (Figs. 4.39 & 4.40). The presence of cellulose and hemicellulose that is not degraded can be explained by the fact that lignin forms a physical barrier around some cellulose and hemicellulose which eliminates microbial access (Tong et al., 1990). The lignin content increased due to a relative decrease in cellulose and hemicellulose contents and enrichment of the remaining solids in lignin.

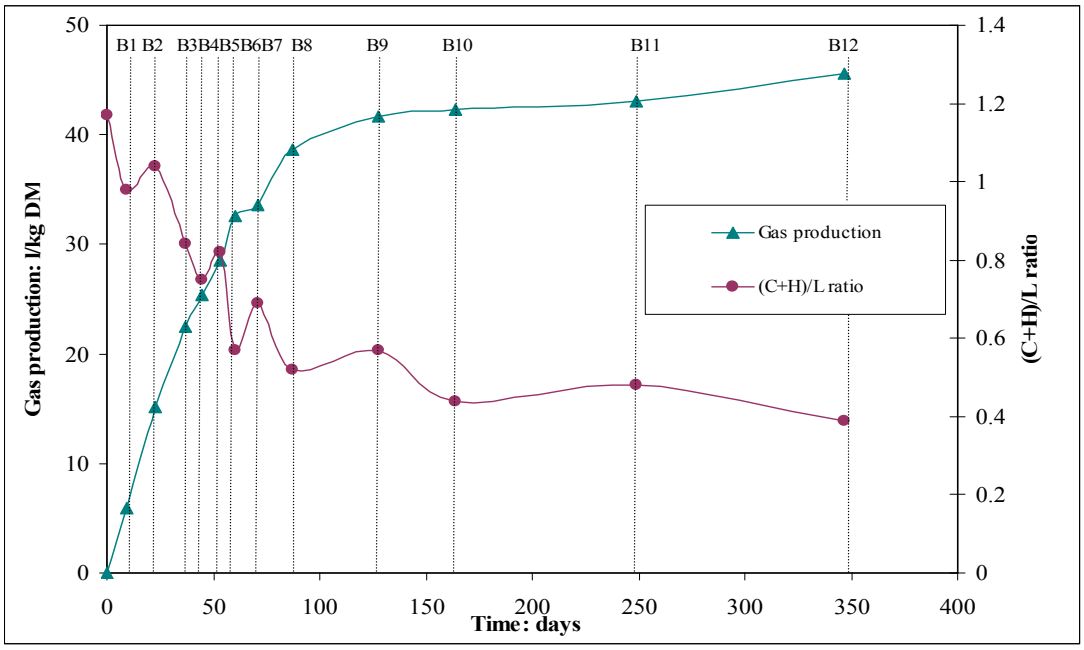


Figure 4.41: (C+H)/L ratio and gas production in BMP reactors for the UK MBT

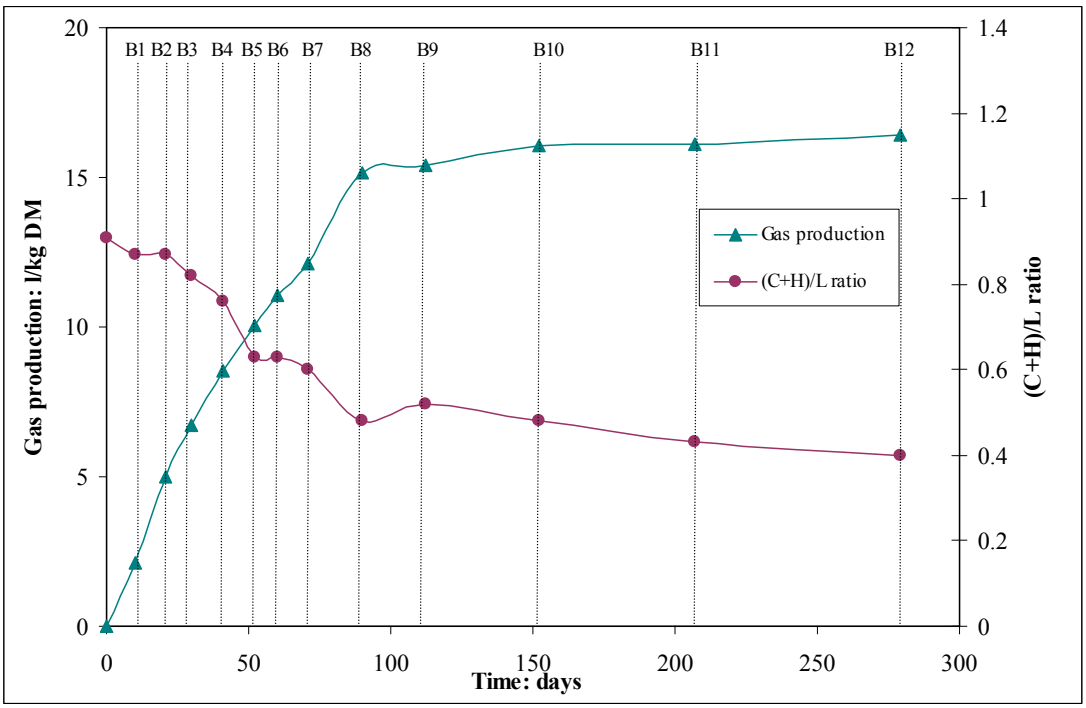


Figure 4.42: (C+H)/L ratio and gas production in BMP reactors for the German MBT waste

A common indicator of the biodegradation potential or the extent of decomposition is the cellulose plus hemicellulose to lignin ratio, (C+H)/L (Wang et al., 1994,; Mehta et al., 2002; Hossain et al., 2003; Zheng et al., 2007). Figures 4.41 and 4.42 show that (C+H)/L ratio decreases with the increasing volume of biogas as decomposition progresses. After 347 days of biodegradation of the UK MBT waste, it had decreased from an initial value of 1.17 to 0.39 due to the decreasing cellulose and hemicellulose content. For the German MBT waste, the (C+H)/L ratio decreased from 0.91 to 0.40. This decrease is in agreement with the data reported by Wang et al., (1994); Hossain et al., (2003) and Zheng et al., (2007). Ivanova et al. (2008*b*) showed a decrease in (C+H)/L ratio from 3.24 to 0.43 during biodegradation of the raw MSW over 919 days. Ivanova et al. (2008*b*) reported a decrease in cellulose

and hemicellulose contents from initial values of about 25% and 7% to about 4% and 3% respectively. Even though the initial cellulose and hemicellulose contents of the pre-treated wastes are reduced, they are found to be very similar for the degraded samples of MSW and MBT wastes indicating that the same state of final decomposition is achieved.

A strong linear correlation was found between the biogas potential (determined from the cumulative biogas yield in the reactors) and the (C+H)/L ratio as shown in Figs 4.43 for the UK and German MBT wastes. Decreasing (C+H)/L ratios correlate well with decreasing biogas potential. The correlation between (C+H)/L ratio and biogas potential for the pre-treated wastes and raw MSW in Fig. 4.44, shows the data on the same line.

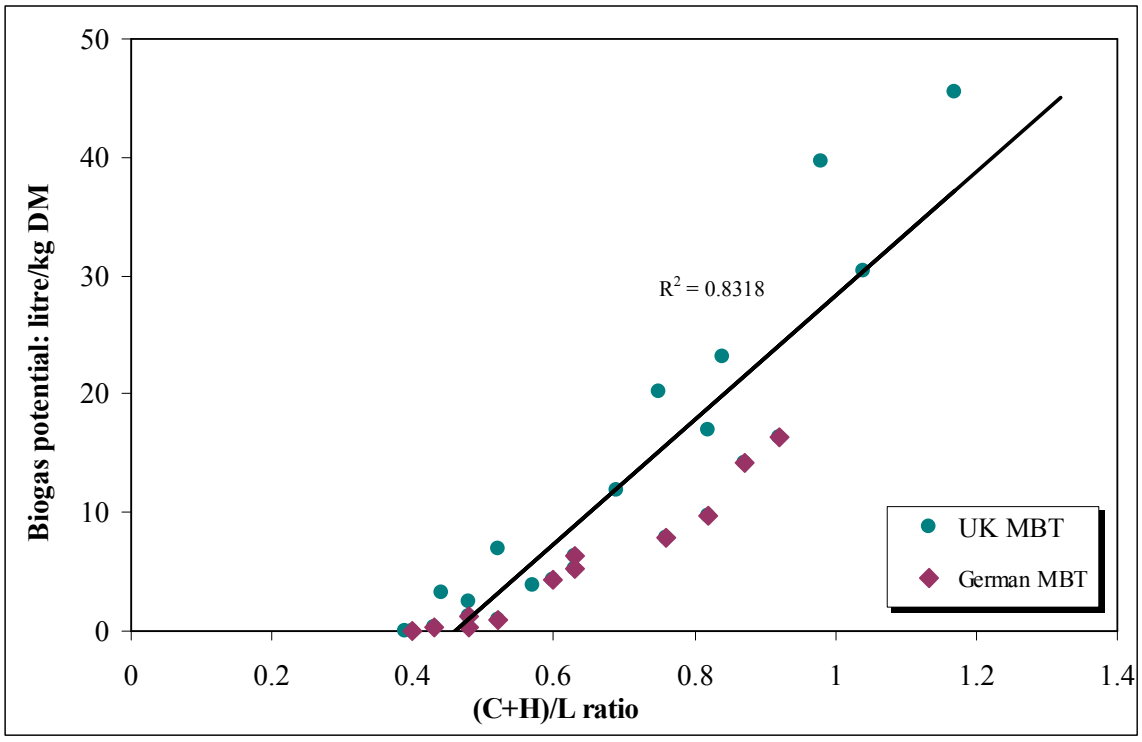


Figure 4.43: Correlation between (C+H)/L ratio and biogas potential for the UK and German MBT wastes

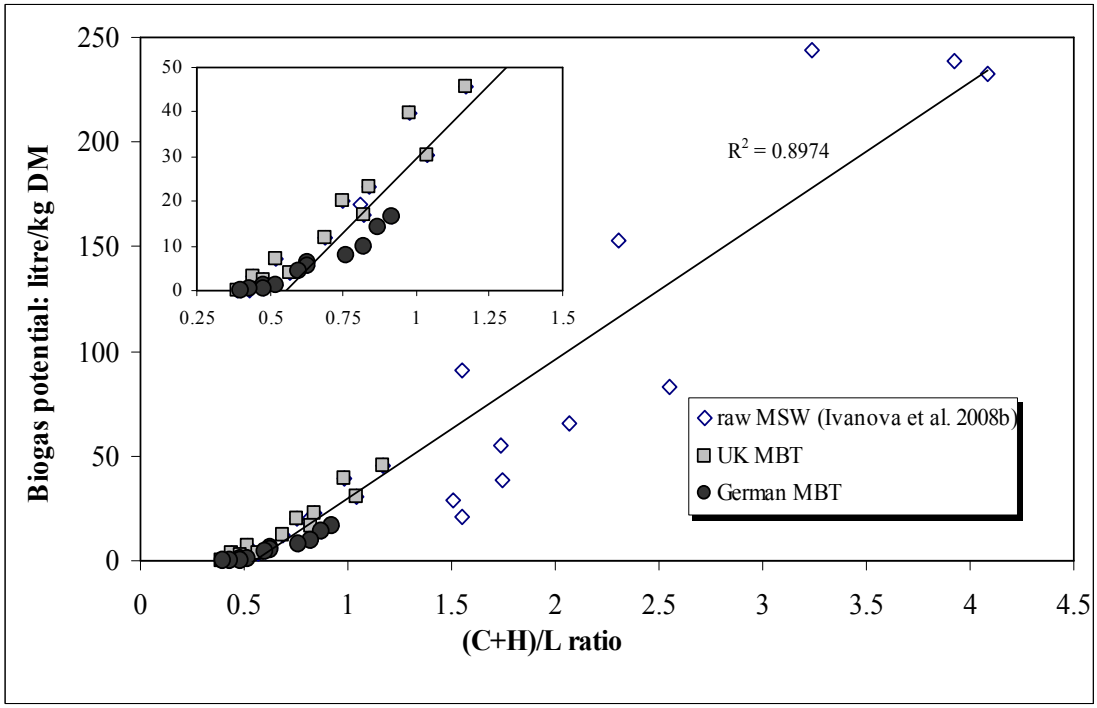


Figure 4.44: Correlation between (C+H)/L ratio and biogas potential for the pre-treated wastes and raw MSW

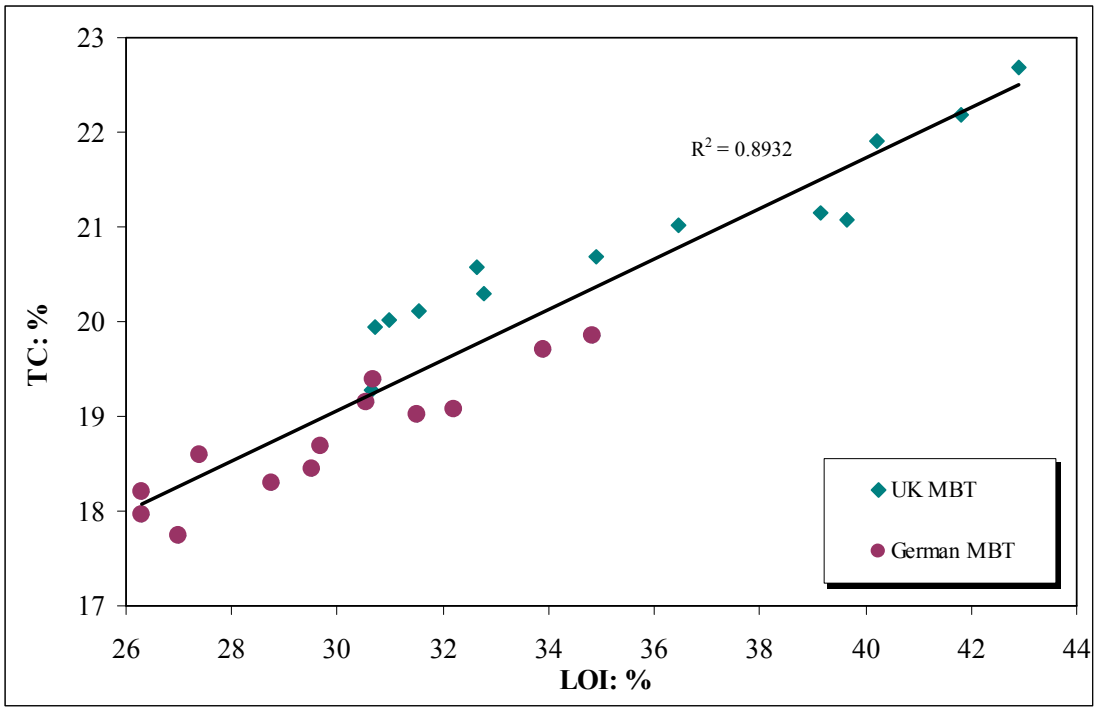


Figure 4.45: Correlation between LOI and TC contents of the UK and German MBT wastes

LOI and TC contents were also used to assess the biodegradation potential of the UK and German MBT residues. Results in Tables 4.1 and 4.2 shows that the LOI content of the waste samples decreased with the progression of degradation which is consistent with the depletion of the TC content. This is probably due to the dependency of TC and LOI on organics in the waste. The LOI content was correlated against TC content (Figure 4.45). LOI content decreased from initial values of 42% and 35% to 30% and 26% after degradation for the UK and German MBT wastes respectively. TC represents the fraction of the waste organic matter determined by LOI content. While the TC includes both organic and inorganic carbon, LOI represents all organic matter including some non-carbon matter (Godley et al., 2004). LOI content includes both degradable (cellulose and hemicellulose) and recalcitrant (lignin, plastic, rubber, leather etc.) organic compounds. For both the UK and German wastes, a good correlation was found when comparing the biogas potential to LOI and TC content of the UK and German MBT wastes (Figs. 4.46 & 4.47). The LOI content of the solid waste samples showed a decreasing trend over the test period, which is consistent with the depletion of TC. Therefore, the changes in solids composition e.g. (C+H)/L ratio, LOI and TC are interrelated and there is a clear link between these parameters and biogas potential.

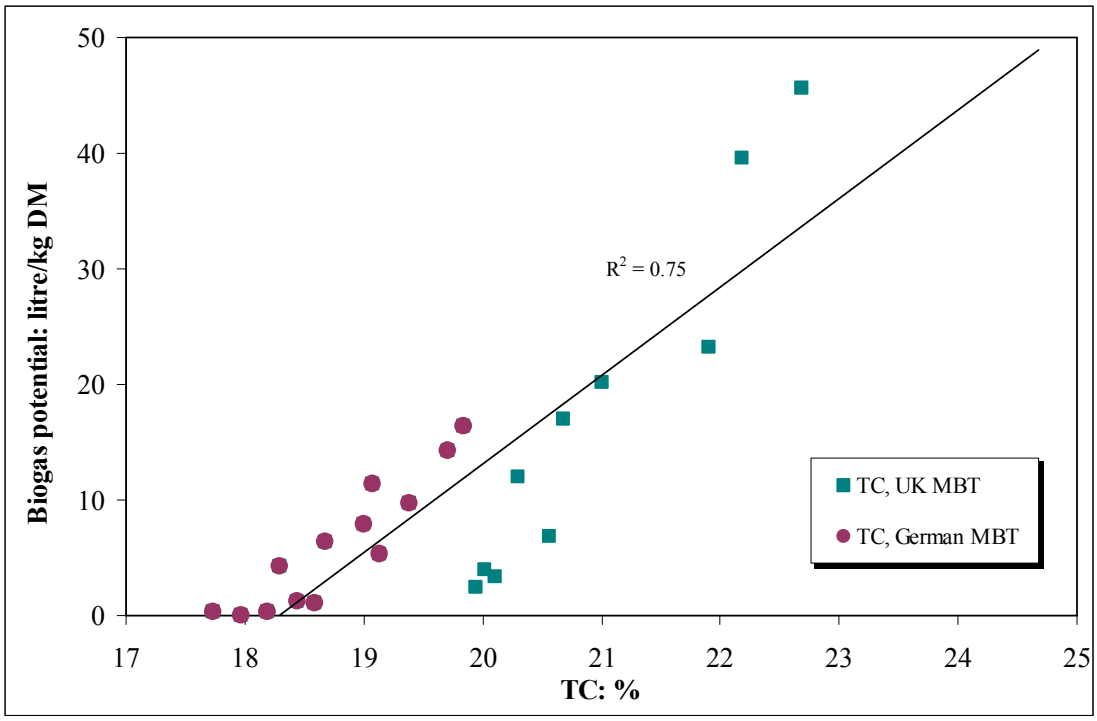


Figure 4.46: Biogas potential correlation with TC for the UK and German MBT wastes

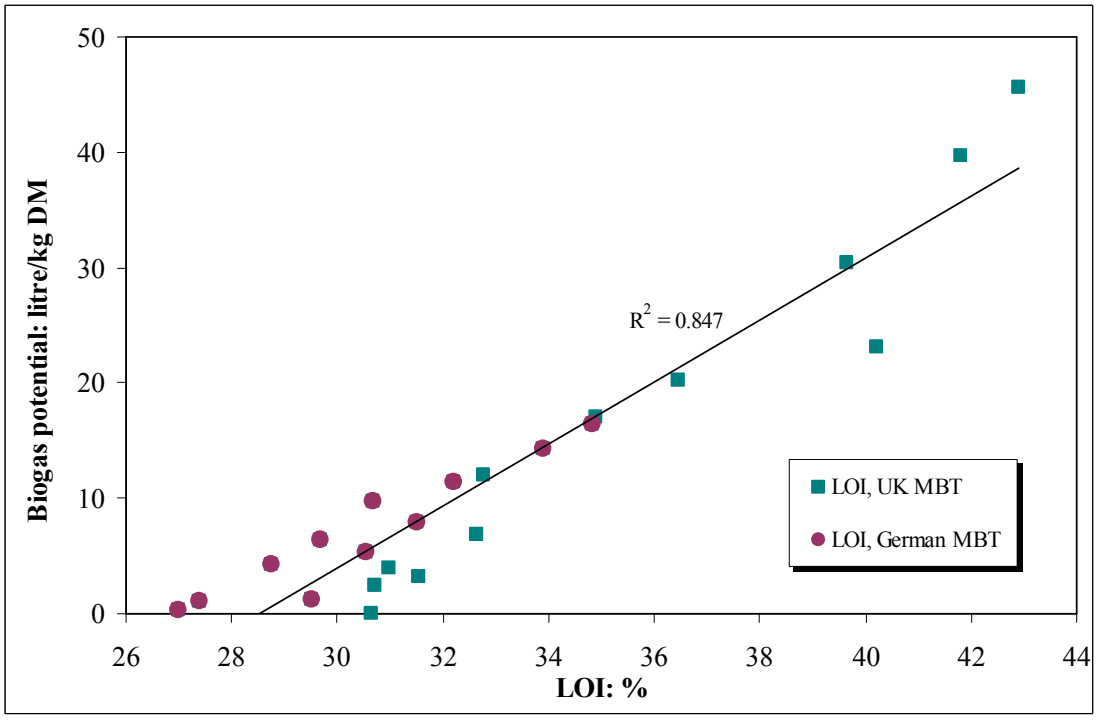


Figure 4.47: Biogas potential correlation with LOI content for the UK and German MBT wastes

4.3 Carbon and nitrogen mass balance analysis

A mass balance confirms quantitatively the effectiveness of a treatment process in environmental engineering. Biodegradation induces a loss of waste constituent mass by transfer to liquid and gas phases and egress from the control volume. Therefore, a mass balance for carbon and nitrogen was made considering the initial and final state of the waste and liquid and gaseous transformations. Carbon and nitrogen mass balances are analysed for the UK and German MBT wastes in CAR2 and in the BMP reactors.

A general mass balance accounting for the overall mass entering (m_{in}), exiting (m_{out}) and accumulating ($m_{accumulated}$) in a system is given by:

$$m_{in} - m_{out} - m_{accumulated} = 0 \qquad \text{Eq. 4.3}$$

The calculations for the carbon and nitrogen mass balance were carried out in an attempt to account for all of the carbon and nitrogen in the system. These calculations provide quantification of the conversion processes involving carbon and nitrogen compounds.

4.3.1 Carbon mass balance

The carbon mass balance was carried out in terms of mass of carbon in grams per kilogram of initial dry mass of the waste. The key components of the carbon mass balance in CAR and BMP reactor that need to be considered are:

1. Carbon Input- This includes the amount of carbon in the waste before the start of test, and the carbon in the synthetic leachate.

(a) Carbon in the initial waste, $C_{initial\ waste}$ (g/kg DM)

$$C_{initial\,waste} = TC_{initial\ waste} \quad \text{Eq. 4.4}$$

where, $TC_{initial\ waste}$ = Total carbon in the initial waste (g/kg DM)

(b) Carbon in the synthetic leachate, $C_{synthetic\ leachate}$ (g/kg DM)

$$C_{synthetic\ leachate} = (TOC + IC)_{synthetic\ leachate} \times \frac{V_{leachate}}{M_{waste}} \quad \text{Eq. 4.5}$$

where, $V_{leachate}$ = volume of leachate (litres)

M_{waste} = initial dry mass of waste (kg)

$(TOC + IC)_{synthetic\ leachate}$ = total organic carbon and inorganic carbon in the synthetic leachate (g/litre)

2. Carbon Output- This includes the amount of carbon in the biogas (methane and carbon dioxide) and leachate produced, and the carbon in the waste after biodegradation.

(a) Carbon in methane, C_{CH_4} (g/kg DM)

$$C_{CH_4} = \frac{[CH_4] \cdot V \cdot Mc \cdot \rho_{CH_4}}{100 \cdot M_{CH_4}} \quad \text{Eq. 4.6}$$

where, $[CH_4]$ = volumetric concentration of methane in biogas (%)

V = cumulative biogas produced (litre/kgDM)

Mc = molar mass of carbon (12 g/mol)

ρ_{CH_4} = density of methane at STP (g/litre)

M_{CH_4} = molar mass of methane (16 g/mol)

(b) Carbon in carbon dioxide, C_{CO_2} (g/kg DM)

$$C_{CO_2} = \frac{[CO_2] \cdot V \cdot Mc \cdot \rho_{CO_2}}{100 \cdot M_{CO_2}} \quad \text{Eq. 4.7}$$

where, $[CO_2]$= volumetric concentration of carbon dioxide in biogas (%)

ρ_{CO_2} = density of methane at STP (g/litre)

M_{CO_2} = molar mass of carbon dioxide (44 g/mol)

(c) Carbon in leachate, $C_{leachate}$ (g/kg DM)

$$C_{leachate} = (TOC + IC)_{leachate} \times \frac{V_{leachate}}{M_{waste}} \quad \text{Eq. 4.8}$$

where, $(TOC + IC)_{leachate}$ = total organic carbon and inorganic carbon in the leachate (g/litre)

(d) Carbon in the degraded waste, $C_{deg\,raded\,waste}$ (g/kg DM)

$$C_{deg\,raded\,waste} = TC_{deg\,raded\,waste} \quad \text{Eq. 4.9}$$

where, $TC_{deg\,raded\,waste}$ = Total carbon in the degraded waste (g/kg DM)

3. Carbon precipitates- The carbon precipitated in the reactor was assumed to be in the form of the precipitates of calcium and magnesium. It is measured from changes in the concentrations of calcium and magnesium in the leachate.

(a) Carbon precipitated as calcium carbonate, C_{CaCO_3} (g/kg DM)

$$C_{CaCO_3} = \frac{[Ca^{2+}] \cdot Mc \cdot V_{leachate}}{M_{Ca} \times M_{waste}} \quad \text{Eq. 4.10}$$

where, $[Ca^{2+}]$=decrease in calcium concentration in the leachate (g/litre)

M_{Ca} = molar mass of calcium (40 g/mol)

(b) Carbon precipitated as magnesium carbonate, C_{MgCO_3} (g/kg DM)

$$C_{MgCO_3} = \frac{[Mg^{2+}] \cdot Mc \cdot V_{Leachate}}{M_{Mg} \times M_{waste}} \quad \text{Eq. 4.11}$$

where, $[Mg^{2+}]$=decrease in magnesium concentration in leachate (g/litre)

M_{Mg} = molar mass of magnesium (24 g/mol)

For a closed system such as CAR and BMP reactor, carbon mass balance can be written as

$$C_{initial\,waste} + C_{synthetic\,leachate} = C_{CH_4} + C_{CO_2} + C_{CaCO_3} + C_{MgCO_3} + C_{leachate} + C_{deg\,raded\,waste} \quad \text{Eq. 4.12}$$

and, mass balance error can be calculated as

Mass balance error (g/kg DM) = True value - Estimated value Eq. 4.13

$$\text{or, Mass balance error}(\%) = \left(\frac{\text{True value - Estimated value}}{\text{True value}}\right) \times 100 \quad \text{Eq. 4.14}$$

where, True value = $C_{\text{initial waste}} + C_{\text{synthetic leachate}}$

and, Estimated value = $C_{CH_4} + C_{CO_2} + C_{\text{leachate}} + C_{\text{deg raded waste}} + C_{CaCO_3} + C_{MgCO_3}$

Carbon mass balance calculations for CAR2 and the BMP reactors over the entire operating period for the UK and German MBT wastes are summarised in Table 4.4. The carbon mass balance for the UK MBT waste indicate that the measured carbon at the end of the experiment accounted for about 96 - 98 % of that initially in the waste: 85 - 86% remained as residual carbon, 10 - 11% was removed in the biogas (about 6-7% methane and 3-4% carbon dioxide), and less than 1% was washed out in the leachate. In the German MBT waste, the measured carbon at the end of the experiment accounted for nearly 95 - 97 % recovery of the initial waste: 90 - 92% residual carbon remaining in the waste, about 4 - 5% biogas (2.5 - 3% methane and 1.5 - 2% carbon dioxide), and less than 0.5% in the leachate. The carbon recovery is similar to those balances proposed by Barlaz et al. (1989*a*), Valencia et al. (2009) and Ivanova et al. (2008*b*). Further, carbon mass balance data revealed that about 70% of the carbon utilised was transformed to biogas whereas only about 10% was retained in the leachate as recalcitrant carbon. The unaccounted carbon fraction might have deposited as carbonates (calcite, siderite etc.) in the drainage layer (Rittman et al. 2003). Additionally, carbon could be lost in the leachate samples and biogas escape during sampling and injection of replaced leachate. Some carbon might have been utilised in bacterial biomass growth.

For the raw MSW, Ivanova et al. (2008*b*) showed that about 70% of the initial carbon was recovered as residual while 26% and 1% was released into the biogas and leachate respectively after degradation for 919 days.

Table 4.4: Summary of carbon mass balance in the UK and German MBT wastes

Parameter[1]	*UK MBT*		*German MBT*	
	BMP	*CAR2*	*BMP*	*CAR2*
$C_{initial\ waste}$, g/kgDM	226.8	226.8	198.5	198.5
$C_{synthetic\ leachate}$, g/kgDM	1.78	0.99	1.49	0.83
C_{CH_4}, g/kgDM	15.13	16.16	5.18	5.42
C_{CO_2}, g/kgDM	9.03	9.54	3.42	3.61
$C_{leachate}$, g/kgDM	2.71	2.5	1.82	1.61
$C_{degraded\ waste}$, g/kgDM	192.7	195.1	179.7	182.9
C_{CaCO_3}, g/kgDM	0.32	0.61	0.28	0.31
C_{MgCO_3}, g/kgDM	0.03	0.04	0.09	0.08
Mass balance error, g/kgDM	8.66	3.84	9.5	5.4
Mass balance error, %	3.8	1.7	4.7	2.7

Note: [1] these parameters were calculated using Equation 4.4 through Equation 4.14

4.3.2 Nitrogen mass balance

The nitrogen mass balance was carried out in terms of mass of nitrogen in grams per kilogram of initial dry mass of the waste. The key components of the nitrogen mass balance in CAR and BMP reactor that need to be considered are:

1. **Nitrogen Input** – This includes the amount of nitrogen in the waste sample before the start of the test, and nitrogen in the synthetic leachate

(a) Nitrogen in the initial waste, $N_{initial\ waste}$ (g/kg DM)

$$N_{initial\ waste} = TN_{initial\ waste} \quad \text{Eq. 4.15}$$

where, $TN_{initial\ waste}$ = Total nitrogen in the initial waste (g/kg DM)

(b) Nitrogen in the synthetic leachate, $N_{synthetic\ leachate}$ (g/kg DM)

$$N_{synthetic\ leachate} = TN_{synthetic\ leachate} \times \frac{V_{leachate}}{M_{waste}} \quad \text{Eq. 4.16}$$

where, $TN_{synthetic\ leachate}$ = Total nitrogen in the synthetic leachate (g/litre)

2. Nitrogen output – This includes the amount of nitrogen in the leachate produced and nitrogen in the waste after biodegradation

(a) Nitrogen output as leachate, $N_{leachate}$ (g/kg DM)

$$N_{leachate} = TN_{leachate} \times \frac{V_{leachate}}{M_{waste}} \quad \text{Eq. 4.17}$$

where, $TN_{leachate}$ = Total nitrogen in the leachate (g/litre)

(b) Nitrogen in the degraded waste, $N_{deg\ raded\ waste}$ (g/kg DM)

$$N_{deg\ raded\ waste} = TN_{deg\ raded\ waste} \quad \text{Eq. 4.18}$$

where, $TN_{deg\ raded\ waste}$ = Total nitrogen in the degraded waste (g/kg DM)

For a closed system such as CAR and BMP reactor, nitrogen mass balance can be written as

$$N_{initial\ waste} + N_{synthetic\ leachate} = N_{leachate} + N_{deg\ raded\ waste} \quad \text{Eq. 4.19}$$

and, mass balance error can be estimated using Equations 4.13 and 4.14.

where, True value = $N_{initial\ waste} + N_{synthetic\ leachate}$

and, Estimated value = $N_{leachate} + N_{deg\ raded\ waste}$

Nitrogen mass balance calculations for CAR2 and the BMP reactors over the entire operating period for the UK and German MBT wastes are summarised in Tables 4.5. Nitrogen mass balances agree within about 5 - 6% for the UK and German MBT wastes. The measured nitrogen at the end of the experiment on UK MBT waste accounted for nearly 94 - 96% recovery: 88 - 89% residual nitrogen and 6 - 7% in the leachate. Similarly, measured nitrogen accounted for about 94 - 95% recovery for the German MBT waste: 92 - 93% residual nitrogen and 2 - 3% in the leachate. Further, nitrogen mass balance data revealed that about 50 – 70% of the nitrogen utilised was retained in the leachate. The nitrogen recovery rates are relatively low since nitrogen determination was not carried out in the gas phase during the experiment. The unaccounted nitrogen might be lost with the gas or leachate samples, and due to microbial uptake or struvite precipitation or several reasons listed in Section 4.1.2.5. Unfortunately, nitrogen mass balance data for the raw MSW is not available for comparison.

Table 4.5: Summary of nitrogen mass balance in the UK and German MBT wastes

Parameter[1]	UK MBT		German MBT	
	BMP	CAR2	BMP	CAR2
$N_{initial\ waste}$, g/kgDM	18.1	18.1	15.2	15.2
$N_{synthetic\ leachate}$, g/kgDM	0.34	0.19	0.24	0.13
$N_{deg\ raded\ waste}$, g/kgDM	16.1	15.9	13.9	14.1
$N_{leachate}$, g/kgDM	1.49	1.24	0.65	0.48
Mass balance error, g/kgDM	0.85	1.15	0.89	0.75
Mass balance error, %	4.6	6.3	5.8	4.9

Note: [1] these parameters are calculated using Equation 4.13 through Equation 4.18

CHAPTER 5 RESULTS AND DISCUSSION-SETTLEMENT & HYDRAULIC PROPERTIES

This chapter presents the results of the investigations into the settlement characteristics of MBT wastes treated to typical UK and German standards. The impacts of creep and biodegradation on the long term settlement were quantified and the data obtained were analysed using simplified settlement models. The results are compared with those of a previous study on raw MSW by Ivanova (2008*a*). Raw MSW data were also analysed using the settlement models. In addition, the hydraulic properties of MBT wastes are examined and discussed.

5.1 Waste Settlement

Waste settlement in the CARs was measured for periods of 347 and 279 days for the UK and German MBT wastes respectively. The CARs were operated at 50 kPa load until day 286 for the UK MBT waste, and until day 202 for the German MBT waste. The load was then increased to 150 kPa for the remainder of the tests to investigate any dependence of the creep characteristics of the waste on the applied load.

During the measurements of drainable porosity and saturated hydraulic conductivity at no applied load, the addition of synthetic leachate to the waste resulted in a settlement of approximately 1%. About 3% settlement was reported by Ivanova (2007) for raw MSW due to liquid addition before

loading. This could be due to the dissolution, transport and/or softening of some of the waste constituents on liquid addition prior to loading.

5.1.1 Immediate Compression

The majority of the settlement occurred instantaneously in response to the application of the load. This immediate compression (measured as percentage of the initial waste height prior to the load application) in the CARs was about 18% -20% for the UK MBT waste and 21-23% for the German MBT waste, and may be attributed to a relative lack of compaction of the waste during the emplacement of waste in CARs. The higher settlements on loading the German MBT waste were probably due to the lesser compaction during filling in CARs to match the initial mass and volume of the UK MBT waste.

5.1.2 Primary Settlement

An arbitrary approach is used by researchers e.g. Chen et al. (2010), Ivanova et al. (2008*a*) for estimating the time for completion of primary settlement for MSW samples tested in laboratories. It seems more reasonable to assess primary settlement using the Terzaghi's theory of one dimensional consolidation, based on the dimensionless parameter

$$T = \frac{c_v t}{d^2} \qquad \text{Eq. 5.1}$$

where T is the time factor; c_v is the consolidation coefficient $c_v = k.E'_o/\gamma_w$, k is the hydraulic conductivity of the waste, E'_o is the constrained modulus $E'_o = \Delta\sigma'_v/\Delta\varepsilon_v$, $\Delta\sigma'_v$ is the increment of vertical effective stress and $\Delta\varepsilon_v$ is the corresponding increment of vertical strain; t is the elapsed time after loading;

γ_w is the unit weight of water and d is the maximum drainage path length. For two-way drainage, the maximum drainage path length d=h/2 for a waste sample of height h.

For simple one dimensional consolidation in response to an increment of total vertical stress, primary settlement is complete when the dimensionless time factor, T in equation 5.1 is about 1 (e.g. Powrie, 2004). The coefficient of consolidation (c_v) was determined using the following two approaches:

(a) Determination of c_v *using graphical approach*

The data of settlement at 50 kPa were plotted against the square root of time in Figs 5.1 and 5.2 for the UK and German MBT respectively. A best fit straight line was drawn through the initial data points and a horizontal line was then drawn to represent an asymptote based on the final data points. The intersection of these lines gives $\sqrt{t_x}$ and c_v is calculated as $c_v = 3d^2/4t_x$ (Powrie, 2004). The values of c_v and T after 24 hours for the UK and German MBT wastes are summarised in Table 5.1.

Table 5.1: Results of consolidation analysis using graphical approach

Parameter	UK MBT waste		German MBT waste	
	CAR1	CAR2	CAR1	CAR2
$\sqrt{t_x}$ (hours)$^{1/2}$	4.3	4.15	4.40	4.45
t_x (seconds)	66564	62001	69696	71289
d (m)	0.247	0.246	0.247	0.247
$c_v = \frac{3d^2}{4t_x}$ (m^2/s)	6.87 x 10^{-7}	7.32 x 10^{-7}	6.57 x 10^{-7}	6.42 x 10^{-7}
T (after 24 hours)	0.97	1.05	0.93	0.91

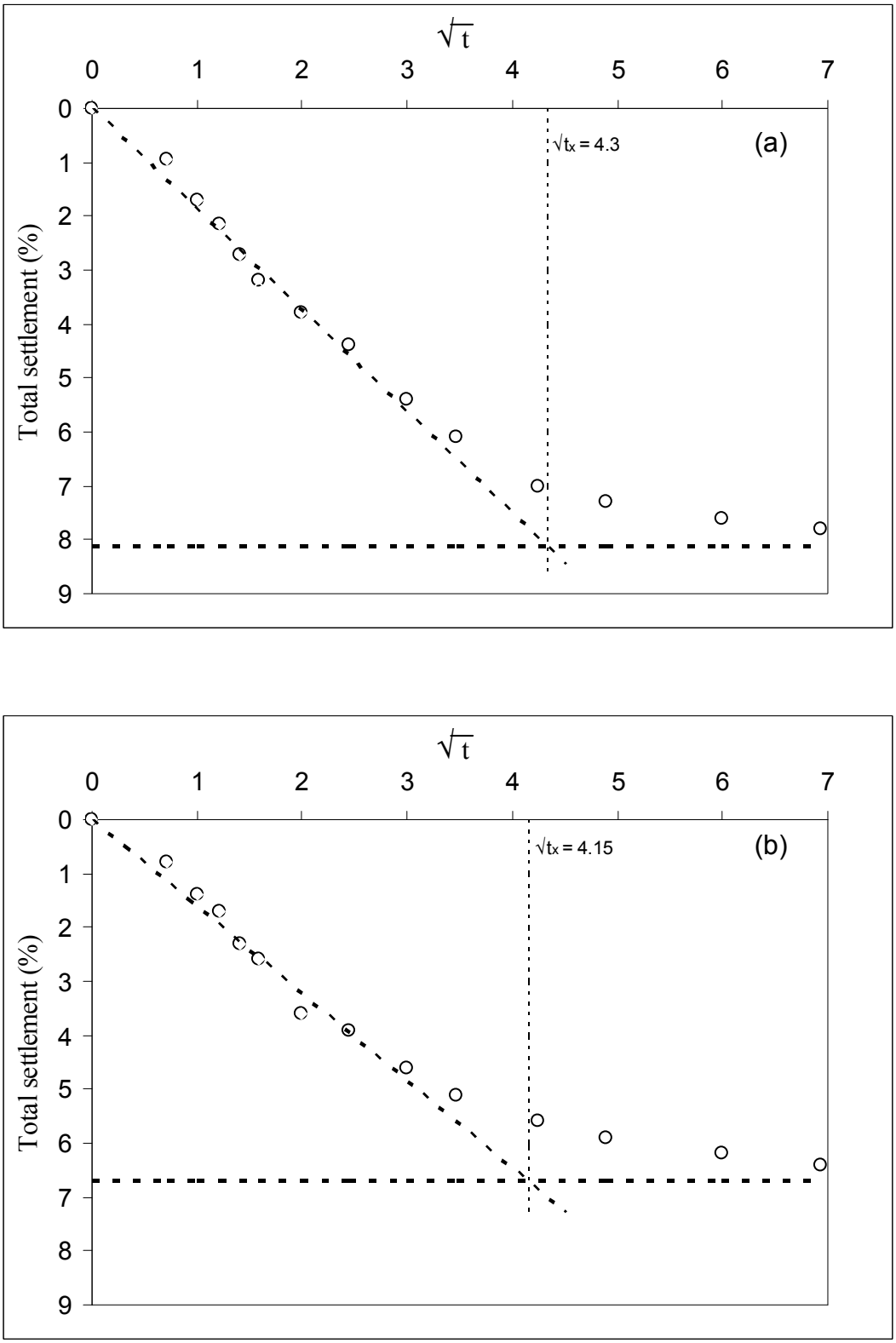


Figure 5.1: c_v determination for the UK MBT waste using graphical approach (a) CAR1 (50 kPa) and (b) CAR2 (50 kPa)

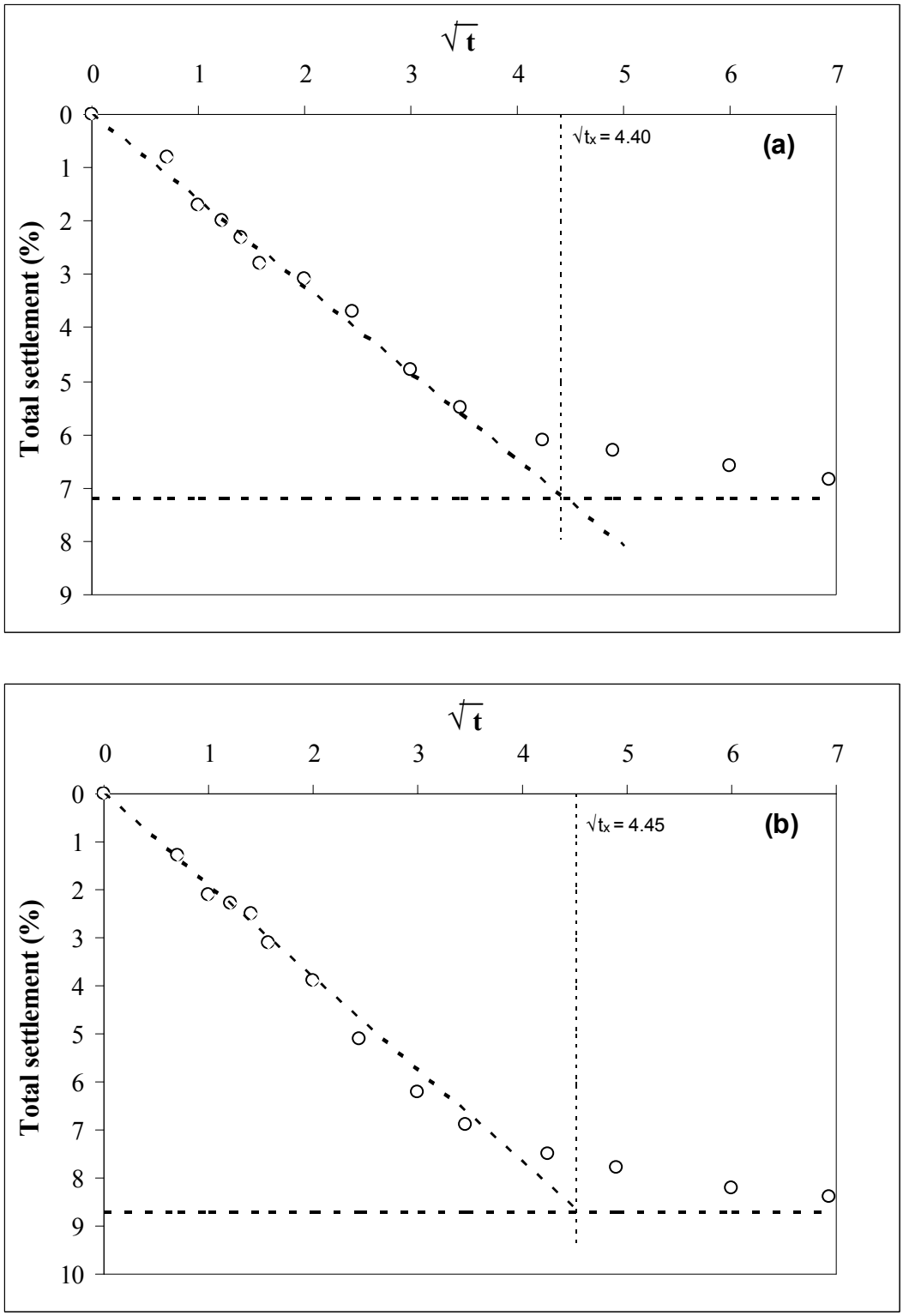


Figure 5.2: c_v determination for the German MBT waste using graphical approach (a) CAR1 (50 kPa) and b) CAR2 (50 kPa)

(b) Determination of c_v *using hydraulic conductivity of the waste*

The value of c_v was estimated indirectly using the relationship

$$c_v = \frac{kE'_o}{\gamma_w} \qquad \text{Eq. 5.2}$$

where $E'_o = \Delta\sigma'_v/\Delta\varepsilon_v$, and $\Delta\varepsilon_v = \Delta h/h$, where Δh is the change in sample height due to an increase in vertical stress of $\Delta\sigma'_v$. The hydraulic conductivity (k) was taken as the average of the values at 0 and 50 kPa loads. The values of c_v and T after 24 hours for the UK and German MBT wastes are summarised in Table 5.2.

Table 5.2: Consolidation analysis (50 kPa loading) using hydraulic conductivity of the waste

Parameter	UK MBT waste		German MBT waste	
	CAR1	CAR2	CAR1	CAR2
$\Delta\sigma'_v$ (kPa)	50	50	50	50
k (m/s)	4.24×10^{-5}	3.85×10^{-5}	3.6×10^{-5}	3.46×10^{-5}
h (m)	0.494	0.492	0.495	0.495
Δh (mm)	121.59	122.49	137.59	134.86
E'_o (kPa)	203.14	200.8	180	183.52
$c_v = \frac{kE'_o}{\gamma_w}$ (m^2/s)	8.77×10^{-7}	7.89×10^{-7}	6.61×10^{-7}	6.45×10^{-7}
T (after 24 hours)	1.24	1.13	0.94	0.92

The values of T after 24 hours in Tables 5.1 and 5.2 are generally around or greater than one, justifying the selection of 24 hours as the time for completion of primary settlement.

Using the hydraulic conductivity of the UK and German MBT wastes, the values of c_v and T for 150 kPa loading stage are calculated and summarised in Table 5.3.

Table 5.3: Consolidation analysis (150 kPa loading) using hydraulic conductivity of the waste

Parameter	UK MBT waste		German MBT waste	
	CAR1	CAR2	CAR1	CAR2
$\Delta\sigma'_v$ (kPa)	100	100	100	100
k (m/s)	3.6×10^{-6}	3.3×10^{-6}	2.3×10^{-6}	1.6×10^{-6}
h (mm)	353.46	338.99	342.78	339.43
Δh (mm)	20.14	20.17	18.48	19.04
E'_o (kPa)	1755.01	1680.7	1854.8	1782.7
$c_v = \frac{kE'_o}{\gamma_w}$ (m^2/s)	6.4×10^{-7}	5.6×10^{-7}	4.3×10^{-7}	3×10^{-7}
T (after 24 hours)	1.7	1.6	1.2	0.91

The primary settlement resulted in additional settlements in the range 5.9% to 7.8% of the waste sample height after immediate settlement for the UK and German MBT wastes. The slight differences between the immediate compression and primary settlement in CAR1 and CAR2 for both UK and German MBT wastes are attributed to small differences in compaction and slight heterogeneity of the wastes. In contrast, total settlements in the range 33% to 48% were observed for the raw MSW (Ivanova et al. 2008*a*) by the end of 24 hours which could be a result of poorer compaction and lower densities (ρ_d =186 kg/m^3 at zero load) of the raw MSW in the CARs. Settlement data for

the UK and German MBT wastes at 50 kPa and 150 kPa in the CARs are summarised in Tables 5.4 and 5.5 respectively.

5.1.3 Secondary Settlement

The secondary settlements measured in the CARs at 50 kPa for the UK and German MBT wastes are plotted against log-time in Figs 5.3 and 5.4 respectively. The secondary settlements are expressed as a percentage of the waste height after primary (consolidation) settlement which was recorded 24 hours following load application. With data collected from both test and control reactors, it was possible to compare and quantify the net effects of creep and biodegradation on settlement. The total secondary settlement was 5.09% in CAR1 and 8.26% in CAR2 for the UK MBT waste, and 4.09% in CAR1 and 5.75% in CAR2 for the German MBT waste.

The rate and magnitude of secondary settlement in CAR2 were higher than CAR1 (shown by the steeper slopes in Figs 5.3 and 5.4) for both the UK and German MBT wastes due to the decomposition of organics, evidenced by the increase in biogas production and decrease in VFA and TOC. The settlement data confirm that the degradation of waste clearly has an impact on the rate and magnitude of settlement. The period of accelerated settlement was followed by period of relatively slow settlement rate showing that the degradation induced settlement was large in the initial stages and then it gradually stabilised with time. After about 200 days for the UK MBT waste and 150 days for the German MBT waste, the settlement rates slowed appreciably in CAR2 indicating that most of the biodegradation had taken place and that biological stability was being approached.

For both wastes, CAR1 remained biologically inhibited over the entire duration of the tests and as a result, settlement may be considered to be due to mechanical creep only. The settlement of the UK and German MBT wastes in CAR2 can be attributed to both mechanical creep and biodegradation. Therefore, settlements due to mechanical creep can be isolated from those due to biodegradation, by comparing the settlements measured in each assuming that creep effects were the same in CAR2 as they were in CAR1.

Secondary settlement associated with mechanical creep was 5.09% and 4.09% and that associated with biodegradation was 3.17% and 1.66% for the UK and German MBT wastes respectively. The mechanical creep component was more significant than the biodegradation induced settlement, which highlights the importance of mechanical creep on the overall magnitude of secondary settlement.

Figs. 5.5 and 5.6 show the secondary settlements expressed as a percentage of the waste height after completion of primary settlement which was recorded 24 hours following an increase in load to 150 kPa, for the UK and German MBT wastes respectively. Mechanical creep continued in the CARs and the settlements are nearly linear with respect to log time exhibiting an approximately constant slope. Gas production in CAR2 for both the wastes was effectively ceased before the increase in load to 150 kPa. CAR1 and CAR2 for the UK and German MBT wastes compressed almost identically in rate and magnitude following the application of the stress of 150 kPa. This indicates that the settlement in the CARs during this stage was entirely dominated by mechanical creep.

Table 5.4: Settlement data for the UK MBT waste in CARs

Reactor	Bulk density kg/m^3	Time period (days)	Settlement (%)	h[1] (mm)	h_i[2] (mm)	h_p[3] (mm)	h_s[4] (mm)	h_2[5] (mm)	h_3[6] (mm)
After leachate addition (for the measurement of n_d and k) prior to loading									
CAR1	679	n/a	1.2	494	n/a	n/a	n/a	n/a	n/a
CAR2	685	n/a	1.6	492	n/a	n/a	n/a	n/a	n/a
After loading at 50 kPa									
CAR1 (50kPa)	940	0 - 1	5.9	n/a	401.74	372.41	n/a	n/a	n/a
CAR2 (50kPa)	949	0 - 1	7.3	n/a	392.68	369.51	n/a	n/a	n/a
At the end of 50 kPa loading (before the increase in load to 150 kPa)									
CAR1 (50kPa)		1 - 286	5.09	n/a	n/a	n/a	353.46	n/a	n/a
CAR2 (50kPa)		1 - 286	8.26	n/a	n/a	n/a	338.99	n/a	n/a
After loading at 150 kPa									
CAR1 (150kPa)		286- 287	5.69	n/a	n/a	n/a	n/a	333.32	n/a
CAR2 (150kPa)		286- 287	5.95	n/a	n/a	n/a	n/a	318.82	n/a
At the end of test at 150 kPa									
CAR1 (150kPa)	1120	287- 347	4.06	n/a	n/a	n/a	n/a	n/a	319.77
CAR2 (150kPa)	1091	287- 347	4.15	n/a	n/a	n/a	n/a	n/a	305.60

Note: [1] h is height of waste after leachate addition prior to loading; [2] h_i is height of waste at instant loading of 50 kPa; [3] h_p is height of waste at the end of primary settlement (24 hours) at 50 kPa; [4] h_s is height of waste at the end of long term secondary settlement at 50 kPa; [5] h_2 is height of waste at the end of primary settlement (24 hours) at 150 kPa; [6] h_3 is height of waste at the end of the test at 150 kPa

Table 5.5: Settlement data for the German MBT waste in CARs

Reactor	Bulk density kg/m^3	Time period (days)	Settlement (%)	h[1] (mm)	h_i[2] (mm)	h_p[3] (mm)	h_s[4] (mm)	h_2[5] (mm)	h_3[6] (mm)
After leachate addition (for the measurement of n_d and k) prior to loading									
CAR1	672	n/a	1.0	495	n/a	n/a	n/a	n/a	n/a
CAR2	677	n/a	1.0	495	n/a	n/a	n/a	n/a	n/a
After loading at 50 kPa									
CAR1 (50kPa)	986	0 - 1	6.2	n/a	381.09	357.41	n/a	n/a	n/a
CAR2 (50kPa)	984	0 - 1	7.8	n/a	390.71	360.14	n/a	n/a	n/a
At the end of 50 kPa loading (before the increase in load to 150 kPa)									
CAR1 (50kPa)		1 - 202	4.09	n/a	n/a	n/a	342.78	n/a	n/a
CAR2 (50kPa)		1 - 202	5.75	n/a	n/a	n/a	339.43	n/a	n/a
After loading at 150 kPa									
CAR1 (150kPa)		202- 203	5.39	n/a	n/a	n/a	n/a	324.3	n/a
CAR2 (150kPa)		202- 203	5.61	n/a	n/a	n/a	n/a	320.39	n/a
At the end of test at 150 kPa									
CAR1 (150kPa)	1140	203- 279	3.61	n/a	n/a	n/a	n/a	n/a	312.59
CAR2 (150kPa)	1145	203- 279	3.75	n/a	n/a	n/a	n/a	n/a	308.36

Note: [1] h is height of waste after leachate addition prior to loading; [2] h_i is height of waste at instant loading of 50 kPa; [3] h_p is height of waste at the end of primary settlement (24 hours) at 50 kPa; [4] h_s is height of waste at the end of long term secondary settlement at 50 kPa; [5] h_2 is height of waste at the end of primary settlement (24 hours) at 150 kPa; [6] h_3 is height of waste at the end of the test at 150 kPa

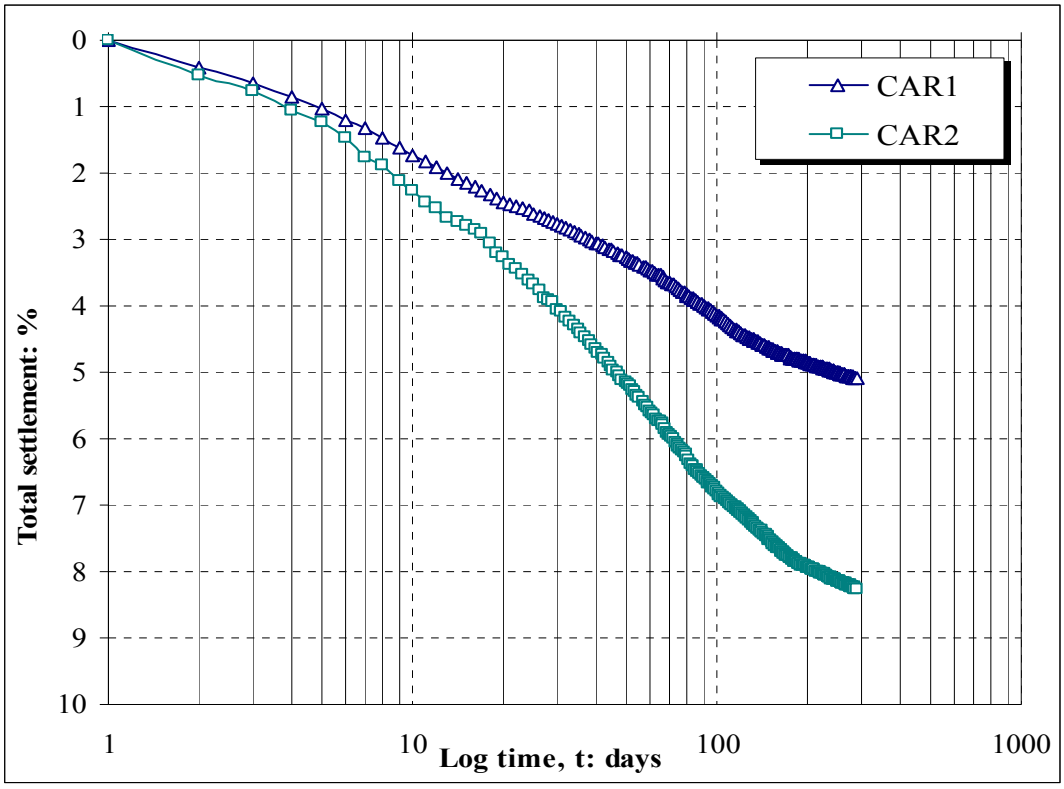


Figure 5.3: Secondary settlement in CARs at 50 kPa for the UK MBT waste

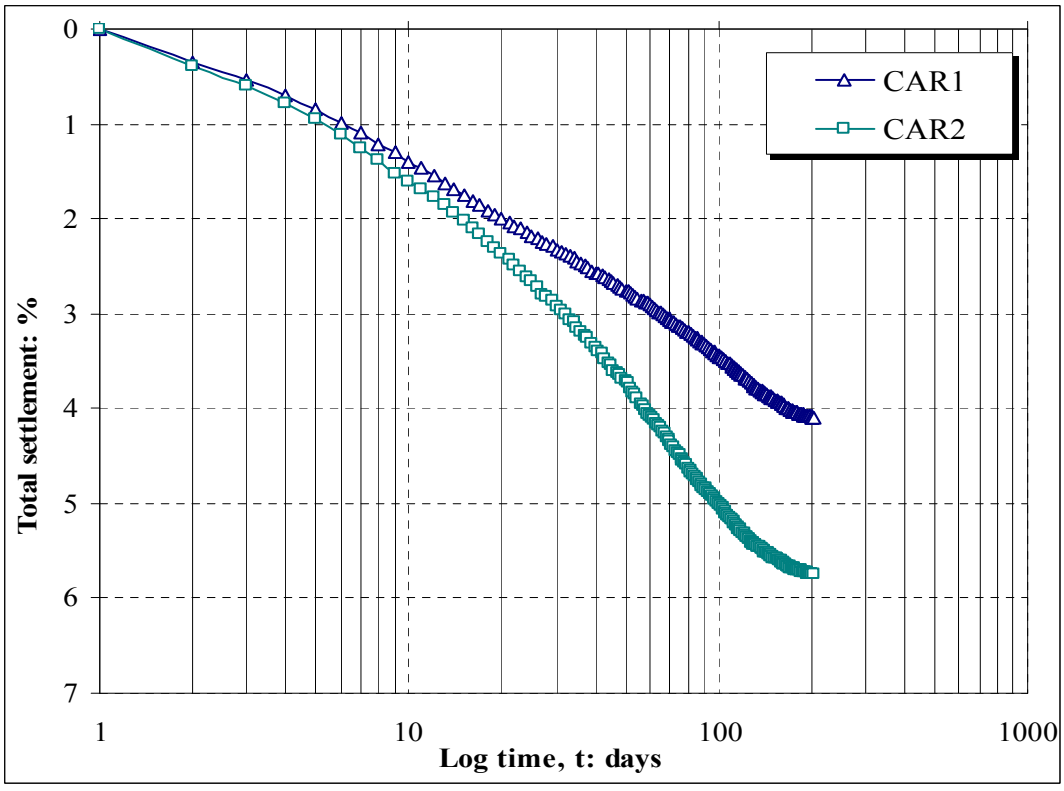


Figure 5.4: Secondary settlement in CARs at 50 kPa for the German MBT waste

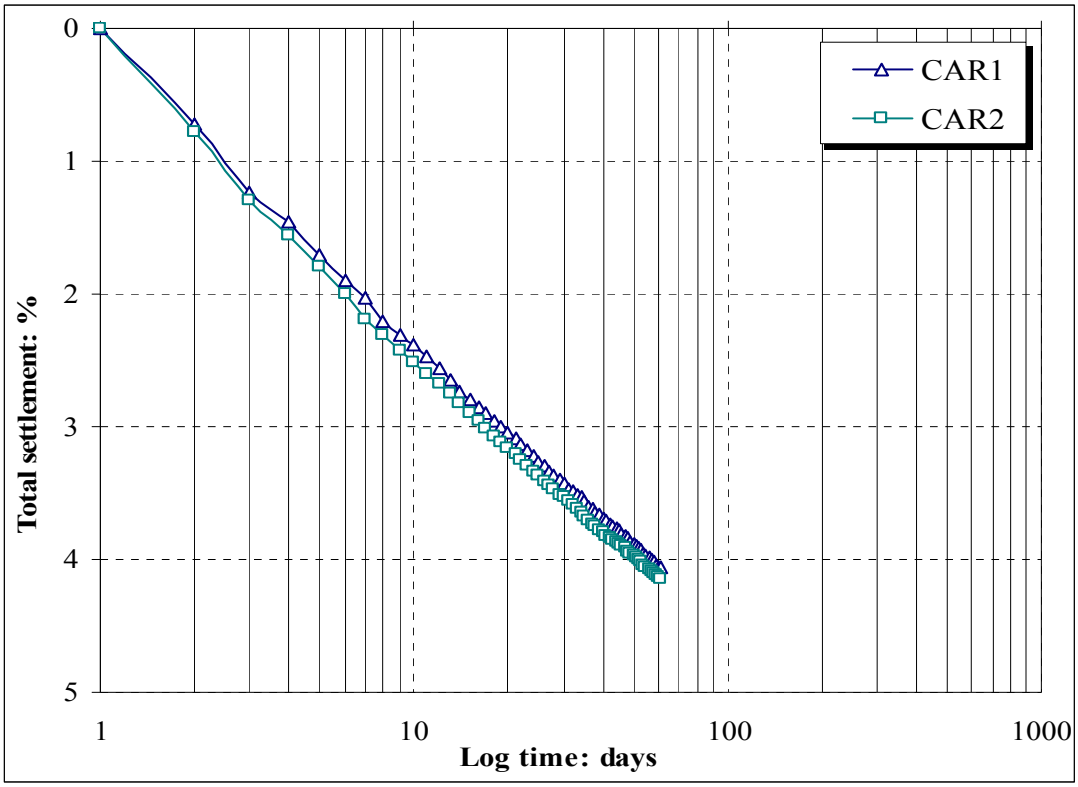


Figure 5.5: Secondary settlement in CARs at 150 kPa for the UK MBT waste

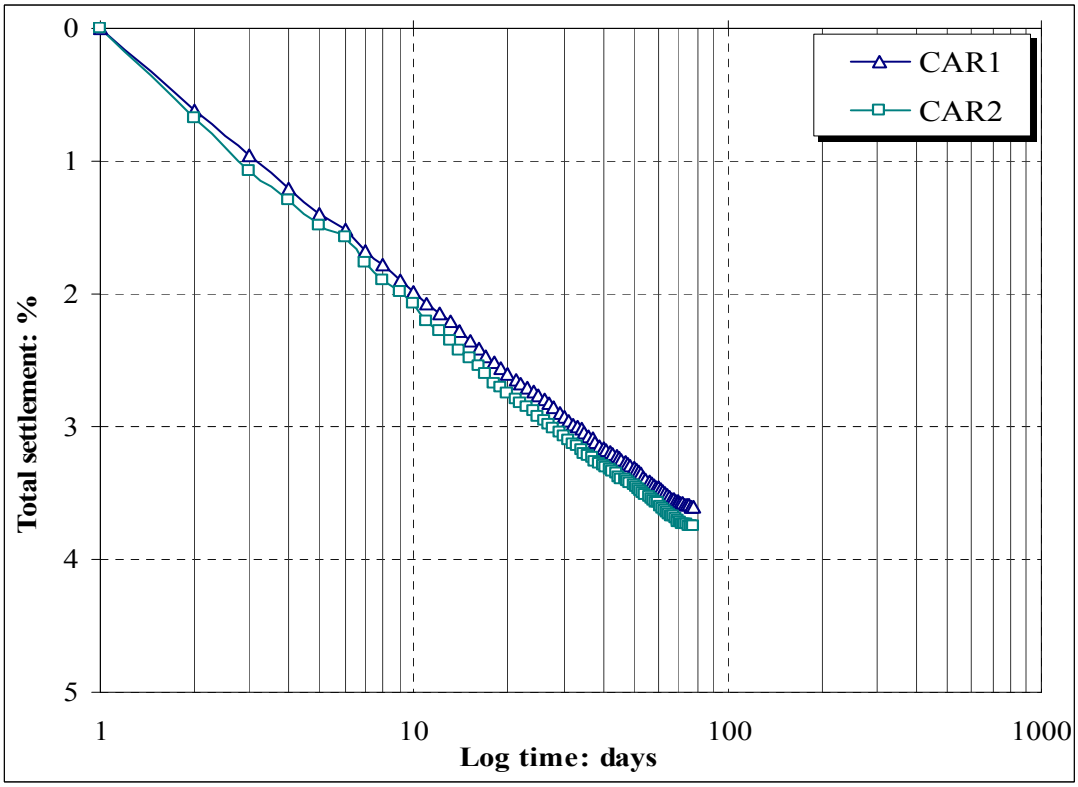


Figure 5.6: Secondary settlement in CARs at 150 kPa for the German MBT waste

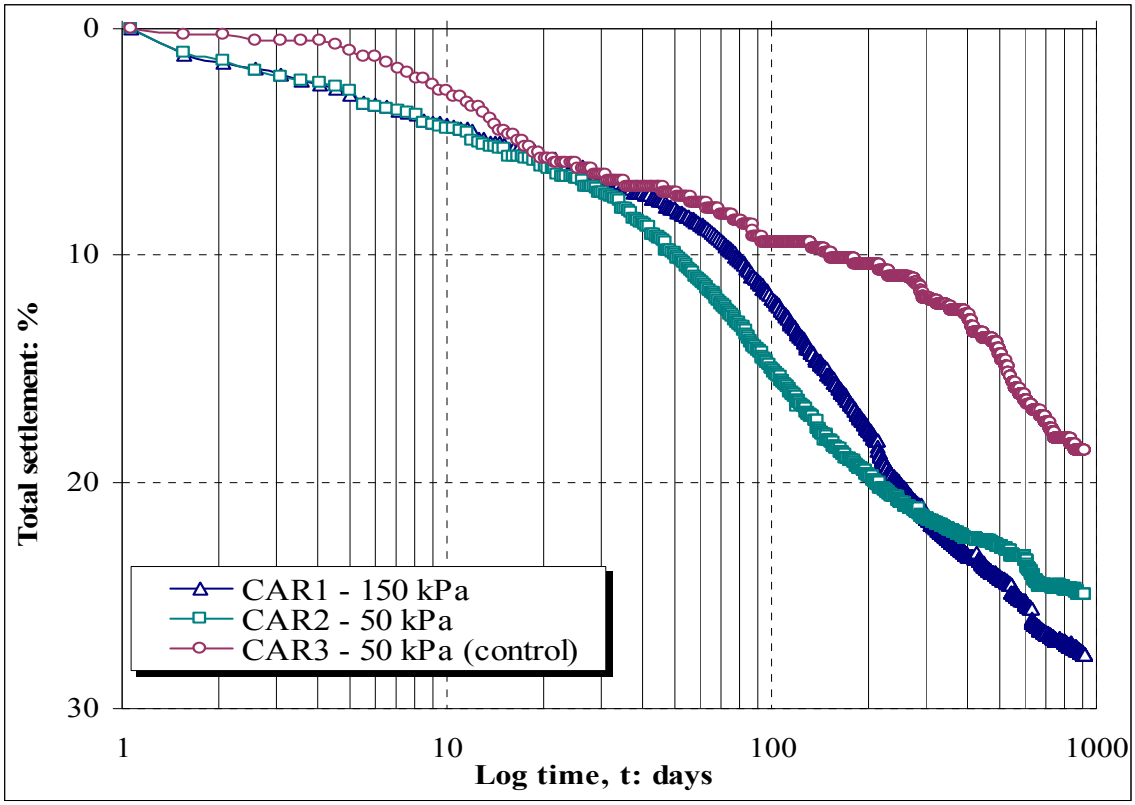


Figure 5.7: Secondary settlement in CARs for raw MSW (after Ivanova et al. 2008a)

Fig 5.7 shows data of secondary settlement against log time from a study on raw MSW in three CARs (CAR1, CAR2 and CAR3) by Ivanova et al. (2008*a*). CAR3 was a control reactor, in which biodegradation was successfully inhibited for the first 314 days of operation, whereas CAR1 and CAR2 were test reactors with enhanced biodegradation at 150 kPa and 50 kPa respectively. The raw MSW was more compressible than the pre-treated wastes. The increase in the rates of compression at about 55 and 32 days correspond to the onset of active biodegradation in CAR1 and CAR2 respectively. The contribution of biodegradation to settlement was established assuming that CAR3 remained inhibited over the entire duration

of the test though it went bioactive and degradation started after 314 days. Creep induced settlement was approximated as a straight line based on the slope of the initial graph of settlement against log-time for CAR3.

Over a period of 919 days, the total creep induced settlement was 13.86% in CAR3 while the biodegradation induced settlements were 13.73% and 11.09% in CAR1 and CAR2 at 50 kPa and 150 kPa respectively. Biodegradation of raw MSW leads to much greater settlement than with pre-treated wastes. This is consistent with the higher biogas production and the high organic content of the raw MSW than the treated material.

Since there is a difference in the duration of tests on the UK MBT waste, German MBT waste and raw MSW, creep induced settlements in control reactors were normalised to two log cycles of time (or 100 days) to compare the values on like to like basis. Table 5.6 presents the relative contributions of mechanical creep and biodegradation induced settlements for the UK and German MBT wastes and compare them with raw MSW.

A slight difference in creep induced settlement values between the UK and German MBT wastes may be explained by difference in waste composition owing to the pretreatment processes and also slight difference in bulk densities of the two wastes.

The biodegradation induced settlement in the German MBT waste was less than in the UK MBT waste. This is consistent with the lower gas production and is a result of low organic content of the German MBT waste.

Table 5.6: Creep and biodegradation induced settlements in CARs at 50 kPa

Parameter	UK MBT waste		German MBT waste		RawMSW (Ivanova et al. 2008*a*)	
	CAR1	CAR2	CAR1	CAR2	CAR2	CAR3
Total secondary settlement, %	5.09	8.26	4.09	5.75	25.0	18.6
Actual creep settlements over actual period of time, %	5.09	5.09	4.09	4.09	13.9	13.9
Estimated creep settlements over 100 days (2 log cycles of time), %	4.11	-	3.52	-	-	9.12
Biodegradation-induced settlement, %	-	3.17	-	1.66	11.1	4.8

Mechanical biological pretreatment of MSW reduces the particle size and eliminates certain components (particularly large and crushable elements) from the waste. Lower values of creep induced settlement in the UK and German MBT wastes compared with the raw MSW could be due to the difference in waste composition owing to the pretreatment and much higher bulk densities of pre-treated wastes.

5.1.4 Correlation between settlement and biodegradation

For the UK and German MBT wastes, creep induced settlement values in CAR1 were subtracted from the settlement values observed in CAR2 to obtain biodegradation induced settlements.

Figure 5.8 shows the relationship of biodegradation between the induced settlements and the amount of biogas produced for the UK and German MBT

wastes. The correlation between the two parameters is close. Ivanova et al. (2008*a*) showed a similarly close relationship between the biodegradation induced settlement and biogas production for the raw MSW in CAR2 at 50 kPa. The data of raw MSW are combined with that of pre-treated wastes and a single correlation is obtained in Fig 5.9. It is perhaps a little surprising that the amount of settlement for a given amount of gas is greater for the MBT waste than for raw MSW, but the difference is small. This difference could also be due to the fact that Ivanova et al. (2008*a*) estimated biodegradation induced settlement based on the assumption that CAR3 remained inhibited of biodegradation throughout the study.

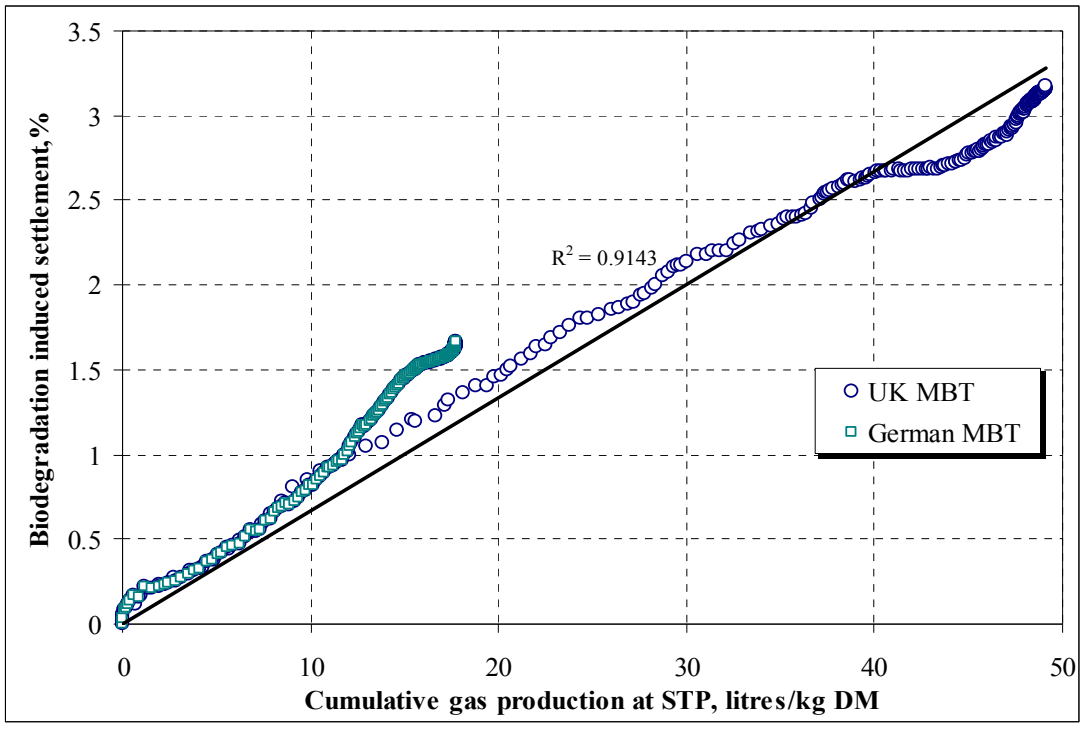


Figure 5.8: Correlation between biodegradation and settlement for the UK and German MBT wastes

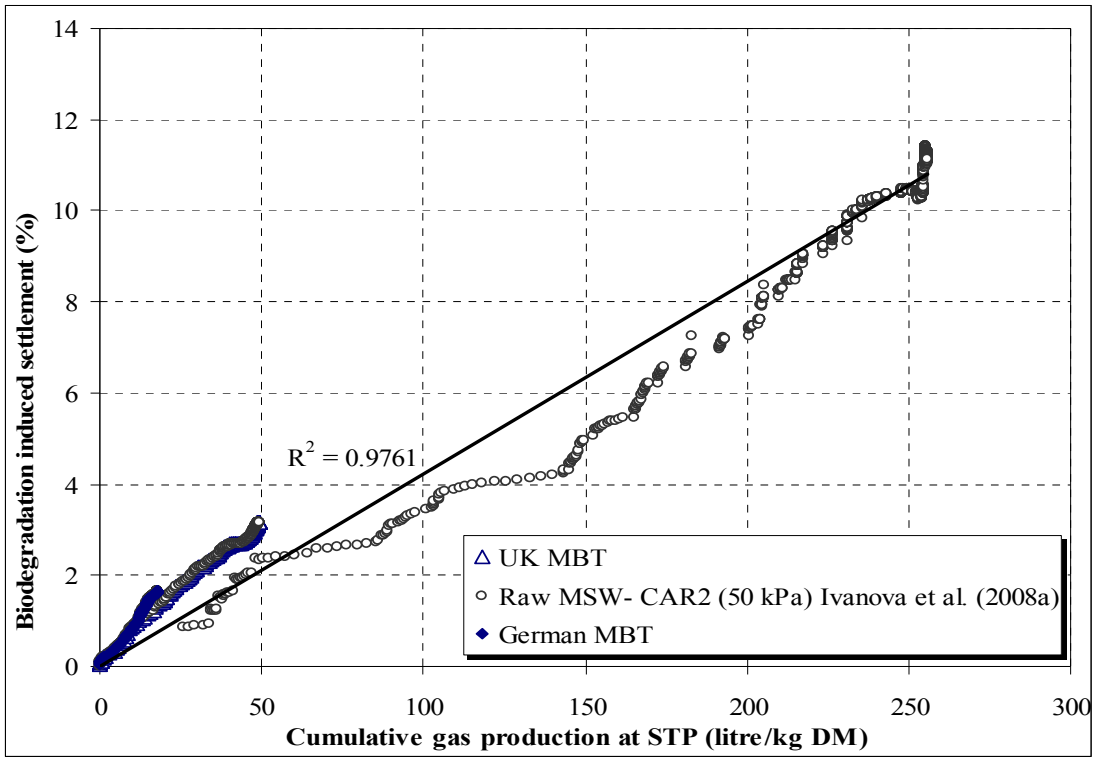


Figure 5.9: Correlation between biodegradation and settlement for the pre-treated wastes and raw MSW

5.2 Waste settlement analysis

The long term settlements associated with mechanical creep and biodegradation were analysed using simplified settlement models. The experimental results of creep and biodegradation induced secondary settlements were used to determine the relevant model parameters. The immediate compression and primary settlement processes were not included in settlement simulations. This approach offers the advantage that the long term settlement parameters obtained can be compared with other similar published data. Also, the models are used as a mathematical tool for the evaluation of the long term settlement behaviour of the waste. The simplified models were applied to data from the current study on MBT wastes and the previous study on raw MSW by Ivanova (2007).

5.2.1 Mechanical creep- induced settlement

Secondary settlement due to mechanical creep can be expressed in simplified form as

$$\varepsilon_c = \frac{\Delta h}{h_p} = C_{\alpha\varepsilon} \log_{10}\left(\frac{t}{t_1}\right) \qquad \text{Eq.5.3}$$

$$\text{or} \quad \varepsilon_c = \frac{\Delta h}{h_p} = \alpha_c \ln\left(\frac{t}{t_1}\right) \qquad \text{Eq. 5.4}$$

where ε_c is the creep induced settlement strain at a time t, Δh is the change in waste height due to creep, h_p is the height of waste at the end of primary settlement, t_1 is the time for completion of primary settlement, $C_{\alpha\varepsilon}$ is the coefficient of secondary settlement and, the creep parameter $\alpha_c = \frac{C_{\alpha\varepsilon}}{2.303}$.

$C_{\alpha\varepsilon}$ describes the change in strain relative to the logarithm (to the base 10) of time. It can be determined from the slope of the graph of settlement data plotted against $\log_{10}$ (time). α_c is the equivalent parameter when natural logarithms are used.

Creep induced settlement strains at 50 kPa in the control reactor (CAR1) for the UK and German MBT wastes are plotted in Figs. 5.10 and 5.11 respectively. The settlements are approximated as linear with respect to the logarithm of time for the determination of $C_{\alpha\varepsilon}$. Settlement strains at 150 kPa plotted against $\log_{10}$ time for the UK and German MBT wastes in CAR1 and CAR2 are shown in Figs. 5.12 and 5.13 respectively. The parameters $C_{\alpha\varepsilon}$ and α_c derived from the settlement data for the UK and German MBT are given in Table 5.7.

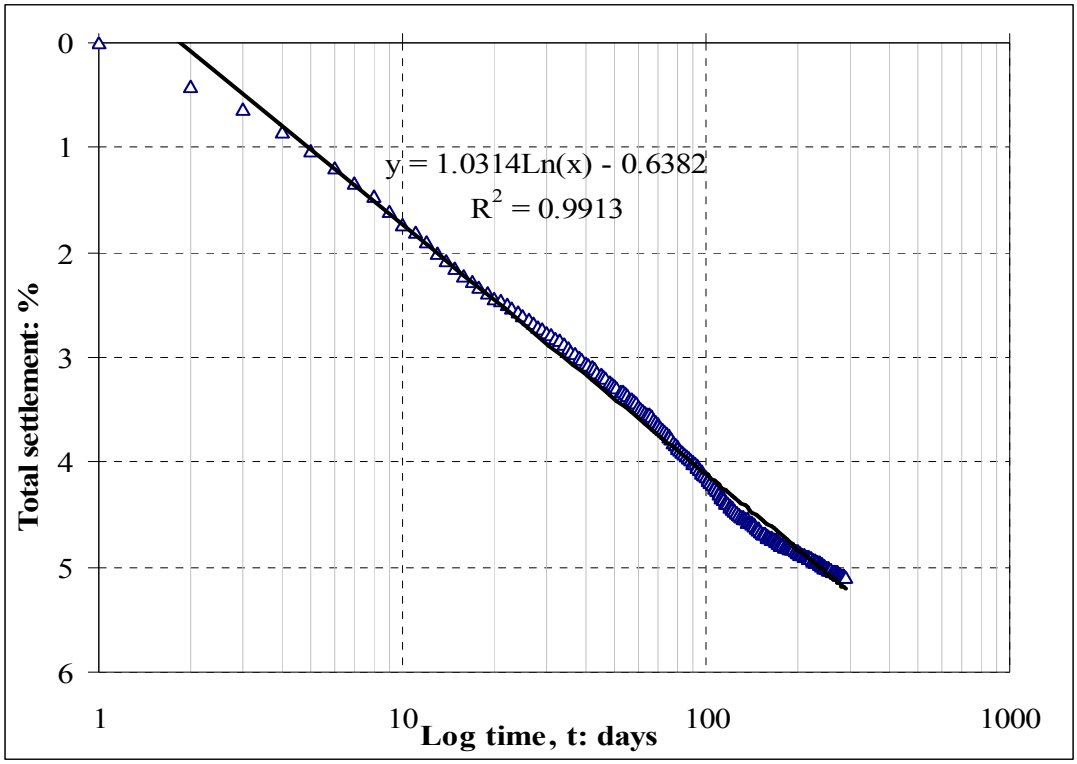


Figure 5.10: $C_{\alpha\varepsilon}$ for the UK MBT waste at 50 kPa using creep settlements in CAR1

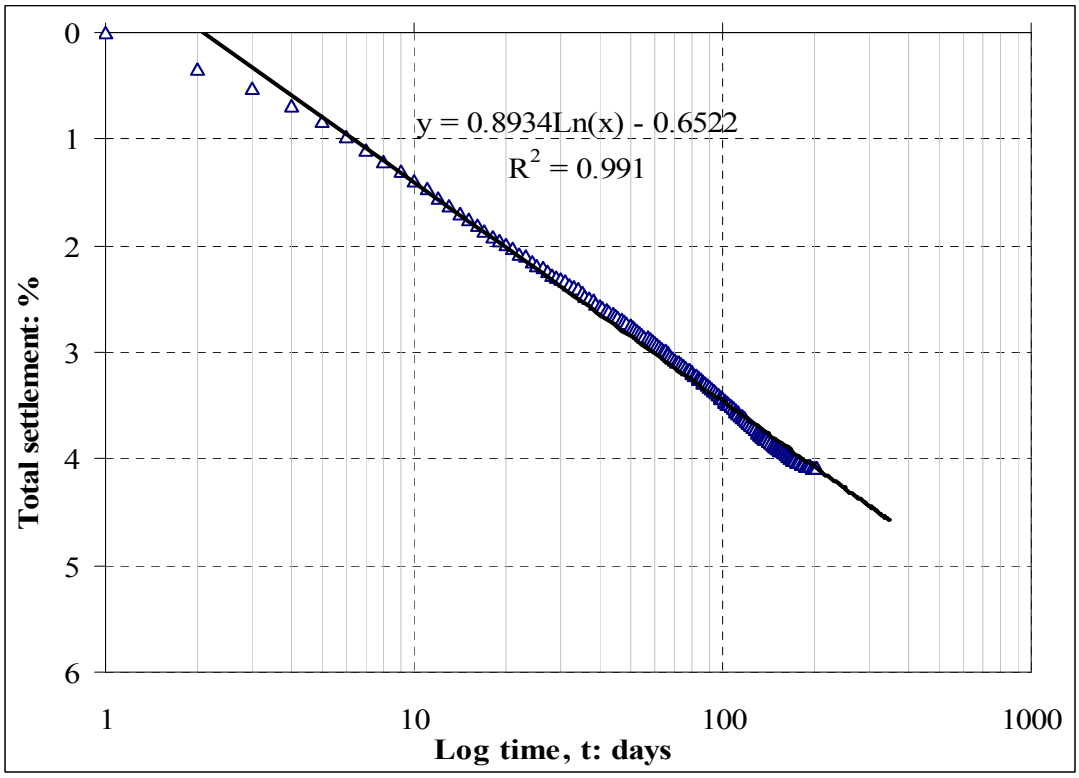


Figure 5.11: $C_{\alpha\varepsilon}$ for German MBT waste at 50 kPa using creep settlements in CAR1

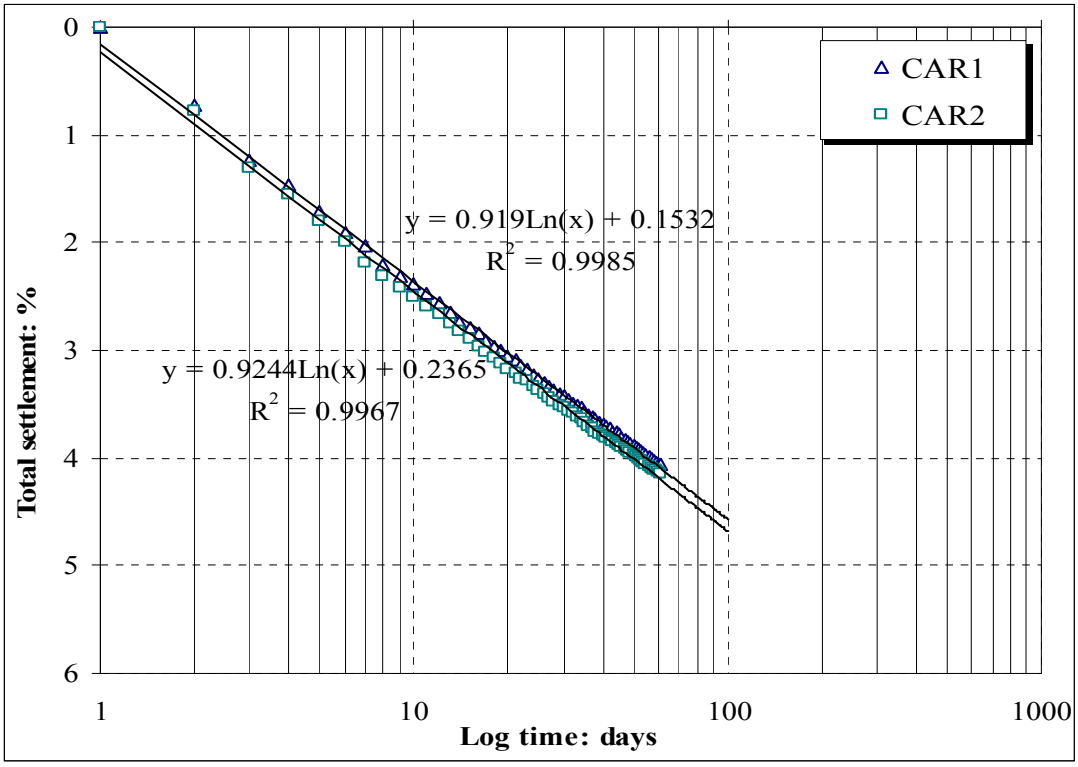


Figure 5.12: $C_{\alpha\varepsilon}$ for the UK MBT waste at 150 kPa in CARs 1 and 2

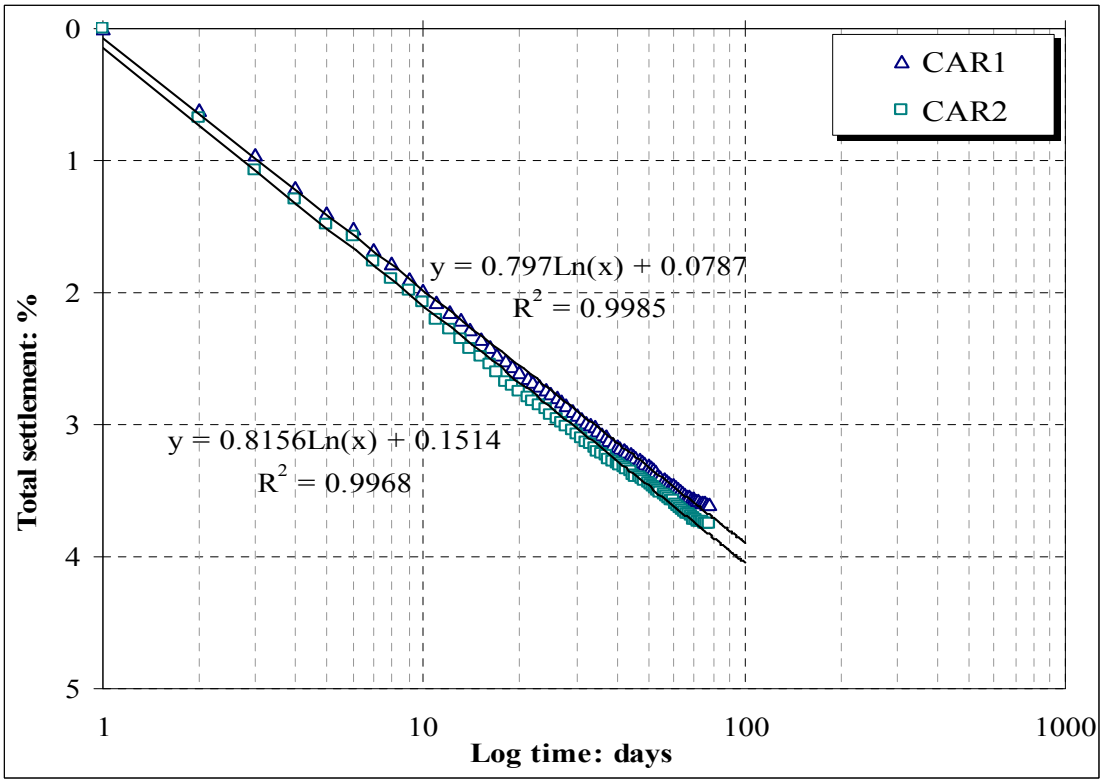


Figure 5.13: $C_{\alpha\varepsilon}$ for the German MBT waste at 150 kPa in CARs 1 and 2

Table 5.7: Creep settlement parameters for the UK and German MBT wastes

Reactor	Bulk unit weight (kN/m^3)	$C_{\alpha\varepsilon}$	Creep parameter, α_c (%)
UK MBT			
CAR1 (50 kPa)	9.4	0.024	1.031
CAR1 (150 kPa)	11.2	0.021	0.919
CAR2 (150 kPa)	10.9	0.021	0.924
German MBT			
CAR1 (50 kPa)	9.9	0.020	0.893
CAR1 (150 kPa)	11.4	0.018	0.797
CAR2 (150 kPa)	11.5	0.019	0.816

There was a slight change in the creep parameter with the increase in applied stress as α_c values for the UK and German MBT wastes were about 10% less at 150 kPa compared with those at 50 kPa. It seems that mechanical creep rate is slightly affected by change in applied stress probably as a result of the accompanying change in density.

Figure 5.14 shows the interpretation of settlement data in the control reactor (CAR3) to establish creep induced settlement for the raw MSW (Ivanova et al. 2008*a*). The initial slope of the graph was used to determine the value of $C_{\alpha\varepsilon}$. Table 5.8 summarises the range of $C_{\alpha\varepsilon}$ and α_c reported in the literature from experiments in which waste degradation was inhibited or did not occur or separated using either a model or the change in slope of the settlement against time.

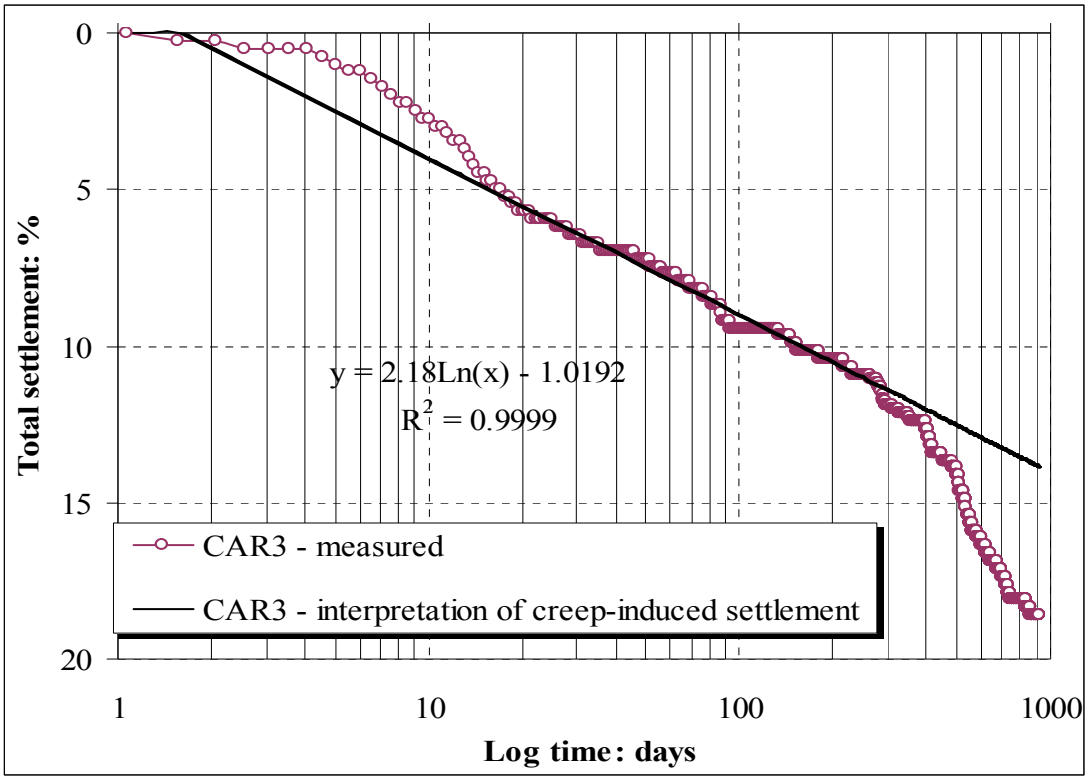


Figure 5.14: $C_{\alpha\varepsilon}$ determination in CAR3 for raw MSW (Ivanova et al. 2008*a*)

$C_{\alpha\varepsilon}$ values for the UK and German MBT wastes were reasonably close and indicative of relatively low settlement rates compared with the range reported in previous studies in Table 5.8. $C_{\alpha\varepsilon}$ was much larger for the raw MSW (Ivanova et al. 2008*a*) than for the pre-treated wastes, which may be attributable to the difference in waste composition and lower bulk density of raw MSW in CARs. This is an interesting point and there could be a possibility that creep rate may depend on bulk density rather than stress.

Table 5.8: Published values of $C_{\alpha\varepsilon}$ and α_c

Reference	Bulk unit weight kN/m³	$C_{\alpha\varepsilon}$	Creep parameter α_c, %	Comments
Hossain et al. (2003) B_1[a]-B_4[a] C_1-C_4		 0.015-0.03 0.02	 0.651-1.302 0.868	C_1-C_4 reactors were neither seeded nor neutralised to retard the onset of methane production. B_1[a]-B_4[a]- inhibited of biodegradation with 6% acetic acid during the compressibility tests
Ivanova et al. (2008)[b]	4	0.05	2.18	Biodegradation was inhibited in CAR3 (50 kPa) for about 350 days
Bareither et al. (2008)	5	0.01-0.05	0.434-2.18	microbiological activity was suppressed using biocide in non biological cell
Rong-Her Chen et al. (2010)	8	0.013-0.04	0.564-1.737	Biodegradation was inhibited by low temperature ~ 8±2°C using a cooling system
Watts and Charles (1999) Calvert landfill site Heathfield landfill site	8	0.02 0.007	0.868 0.304	Settlement monitoring at MSW landfill sites over a period of about 10 years
Park and Lee (2002) Fresh waste 15-30 years old waste		 0.024-0.037 0.008-0.01	 1.042-1.602 0.347-0.434	Applied their model to obtain creep parameters from previous published data of Sowers (1973), Rao et al.(1977), Jessberger and Kochel (1993) and Gabr and Valero (1995)
Chen et al. (2010)	6.3 12.1	0.012 0.0071	0.521 0.308	Biodegradation was inhibited using Na_2CO_3 and low temperature~2-10°C
Benson et al. (2007)[c]		0.026-0.029	1.13-1.26	Settlement monitoring at landfill sites-creep was distinguished from biodegradation induced settlements by slope of settlement –logt curve for the first 500 days of operation
Zacharof and Coumoulos (2001)		0.014-0.026	0.608-1.13	Control cell of Mountain View Landfill Project was analysed for creep by splitting total settlement into two curves distinguished by slopes

Note: [b] from Powrie et al. (2009), [c] from Bareither et al. (2010).

5.2.2 Biodegradation- induced settlement

Secondary settlement associated with degradation of waste organics was analysed using a simplified model based on first order reaction kinetics (Park and Lee, 1997, 2002; Elagroudy et al., 2008) in the form

$$\varepsilon_b = \varepsilon_{bt}\left(1 - e^{-k_b t'}\right) \qquad \text{Eq. 5.5}$$

where ε_b is the biodegradation-induced settlement strain at a time t′ after the start of degradation, ε_{bt} is the total settlement strain resulting from biodegradation, and k_b is a degradation rate constant. The model rests on two basic parameters, ε_{bt} and k_b. The value of k_b can be estimated after fixing the value of ε_{bt} obtained from the study. A linear regression approach was used to fit the mathematical model (Eq. 5.5) to the experimental data to find the degradation rate constant k_b.

Eq. 5.5 can be rearranged as

$$\frac{\varepsilon_b}{\varepsilon_{bt}} = \left(1 - e^{-k_b t'}\right) \qquad \text{Eq. 5.6}$$

$$\text{or} \quad e^{-k_b t'} = \left(1 - \frac{\varepsilon_b}{\varepsilon_{bt}}\right) \qquad \text{Eq. 5.7}$$

$$\text{or} \quad -k_b t' = \ln\left(\frac{\varepsilon_{bt} - \varepsilon_b}{\varepsilon_{bt}}\right) \qquad \text{Eq. 5.8}$$

Finally, the equation can be written as

$$\ln\left(\varepsilon_{bt} - \varepsilon_b\right) = -k_b t' + \ln \varepsilon_{bt} \qquad \text{Eq. 5.9}$$

which is an equation of a straight line and k_b can be determined by plotting the data $\ln(\varepsilon_{bt} - \varepsilon_b)$ versus t′ and fitting the linear curve. Biodegradation induced settlements were calculated by subtracting creep settlements (CAR1) from the total settlements (CAR2) for the UK and German MBT wastes.

For the UK MBT waste, k_b values were calculated for three different plausible ε_{bt} values (3.2%, 3.3% and 3.4%) to analyse the sensitivity of k_b to the assumed ε_{bt} (Fig. 5.15). Table 5.9 shows the k_b values corresponding to each of the assumed ε_{bt} values. The results show that k_b is slightly influenced by ε_{bt}, when ε_{bt} is varied within plausible limits.

Table 5.9: Sensitivity analysis of k_b

ε_{bt} (%)	3.2	3.3	3.4
k_b (day^{-1})	0.016	0.014	0.012

Using the experimental results of the current study and the previous study on raw MSW (Ivanova et al. 2008*a*), k_b values were determined for the UK and German MBT wastes (Figs. 5.16 and 5.17), and for the raw MSW in CAR1 and CAR2 (Figs. 5.18 and 5.19). For the UK and German MBT wastes, biodegradation started from the 4th day while for the raw MSW biodegradation started from day 55 and day 32 for CAR (150 kPa) and CAR2 (50 kPa) respectively. The parameters k_b and ε_{bt} derived from the settlement data for the UK MBT waste, German MBT waste and raw MSW are given in Table 5.10. The effect of pretreatment on settlement is reflected in the parameters ε_{bt} and k_b. Table 5.11 summarises the range of k_b and ε_{bt} reported in the literature for various real and simulated wastes.

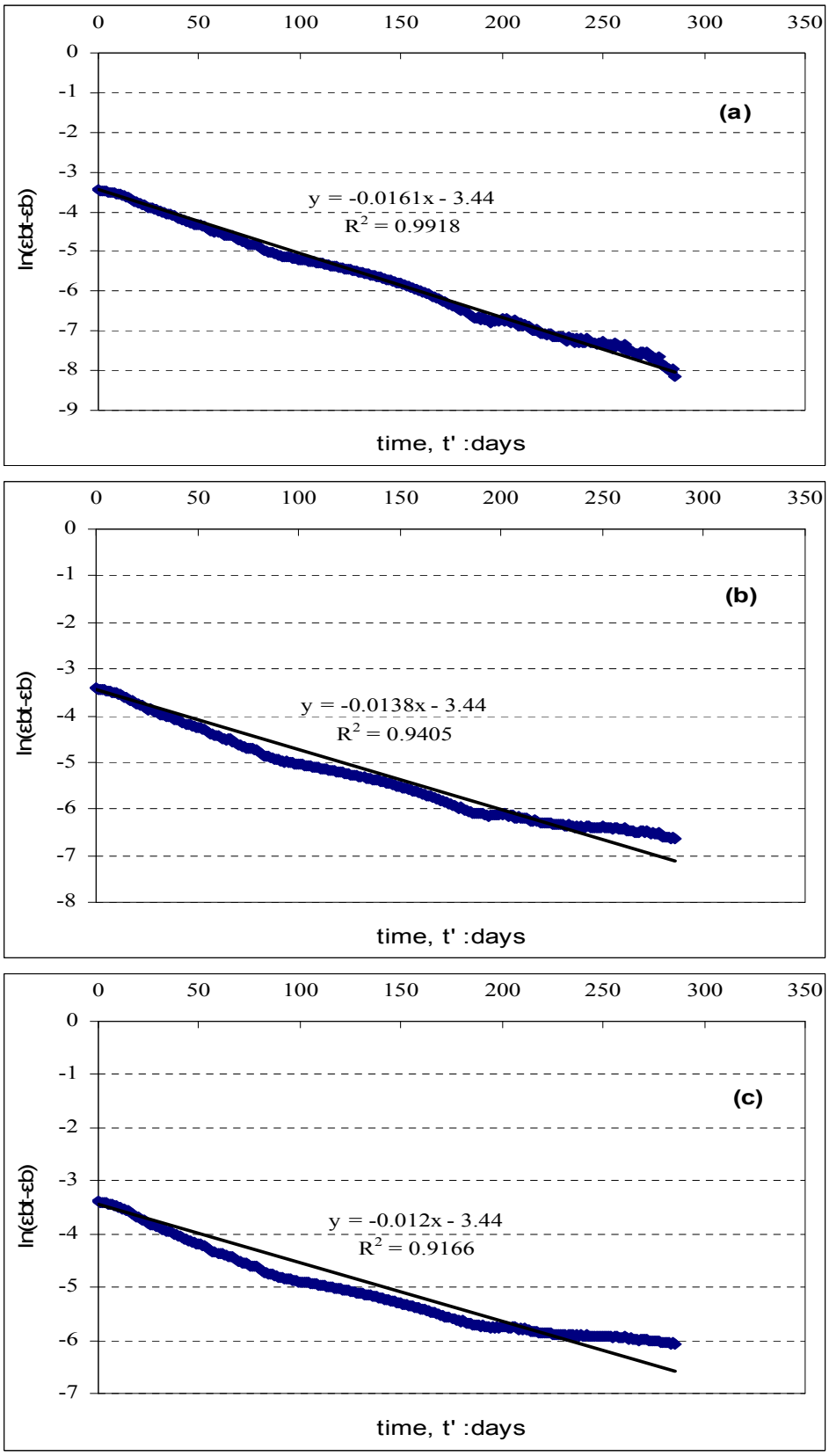


Figure 5.15: Sensitivity of k_b to ε_{bt} : (a) ε_{bt} =3.2, (b) ε_{bt} =3.3 (c) ε_{bt} =3.4

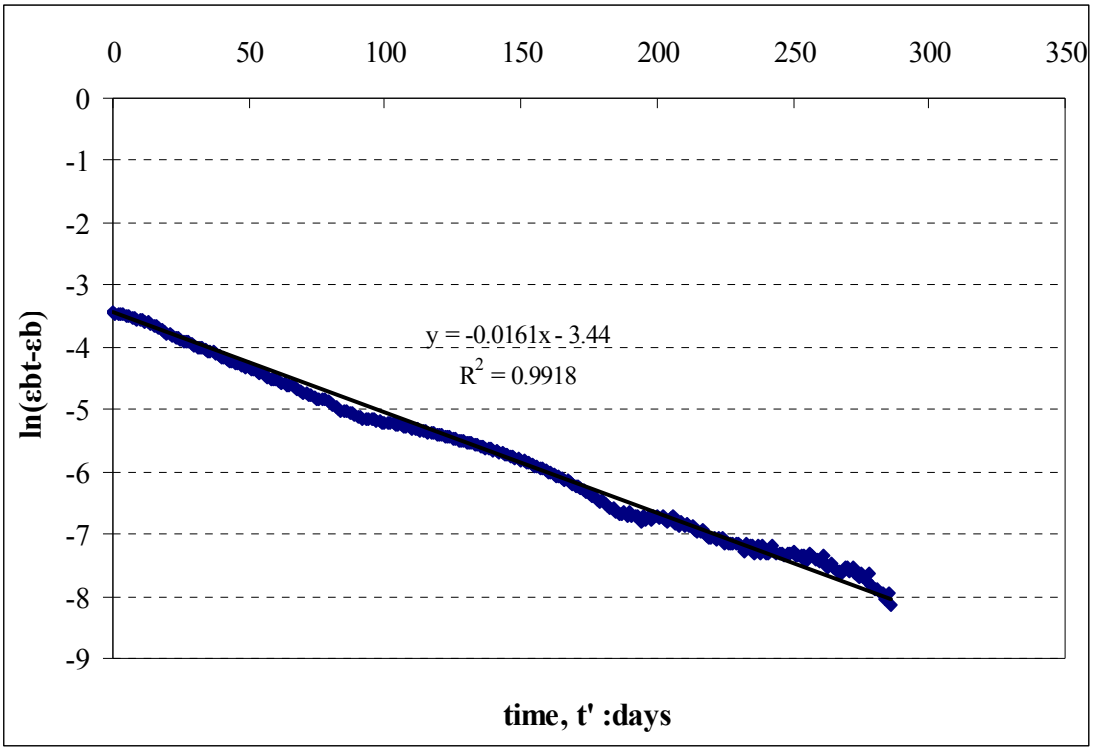


Figure 5.16: Determination of k_b for the UK MBT waste

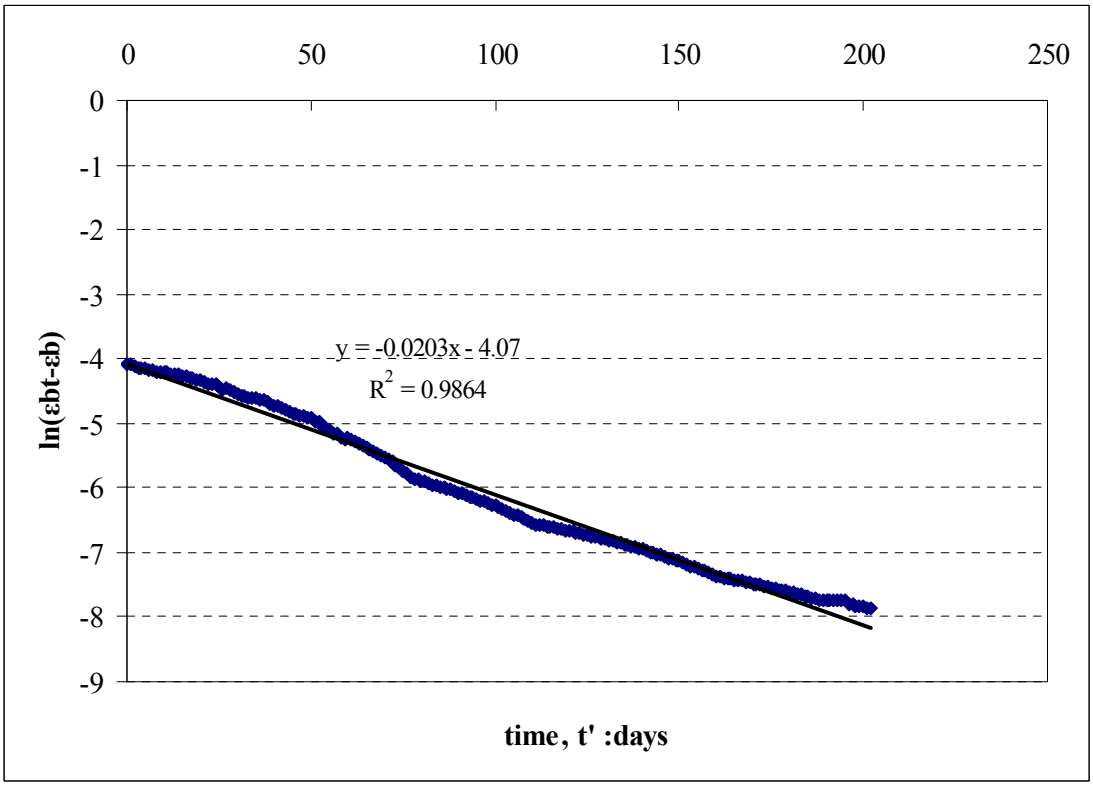


Figure 5.17: Determination of k_b for the German MBT waste

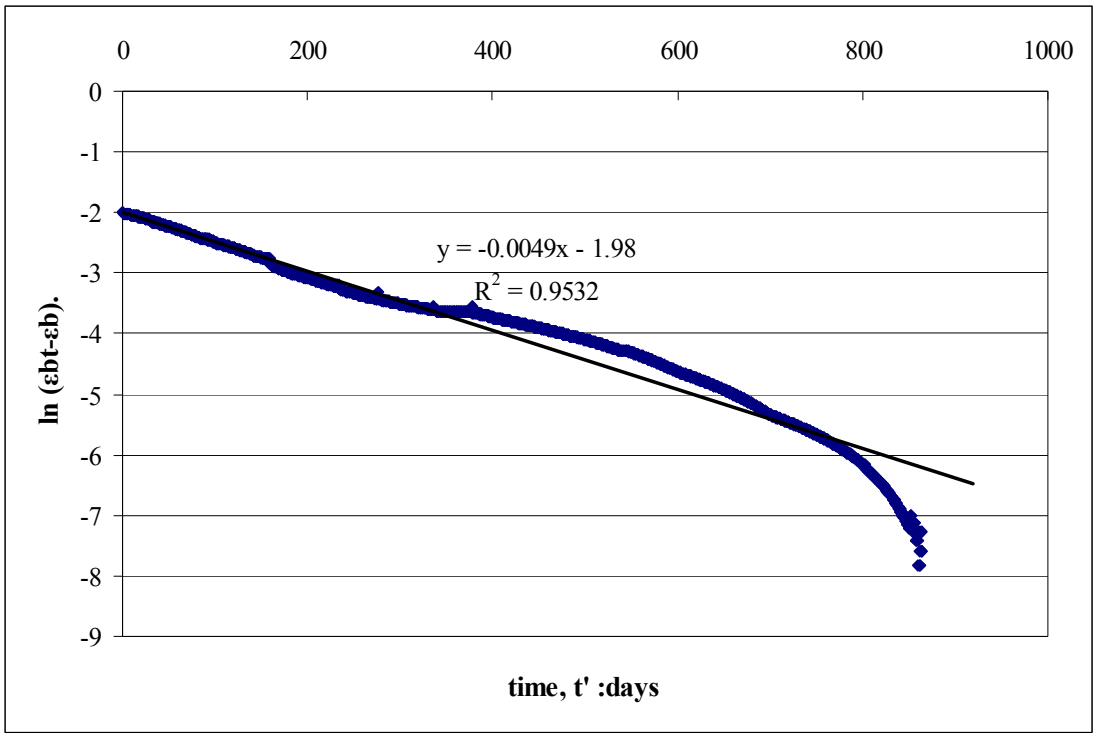


Figure 5.18: Determination of k_b for the raw MSW in CAR2 at 50 kPa

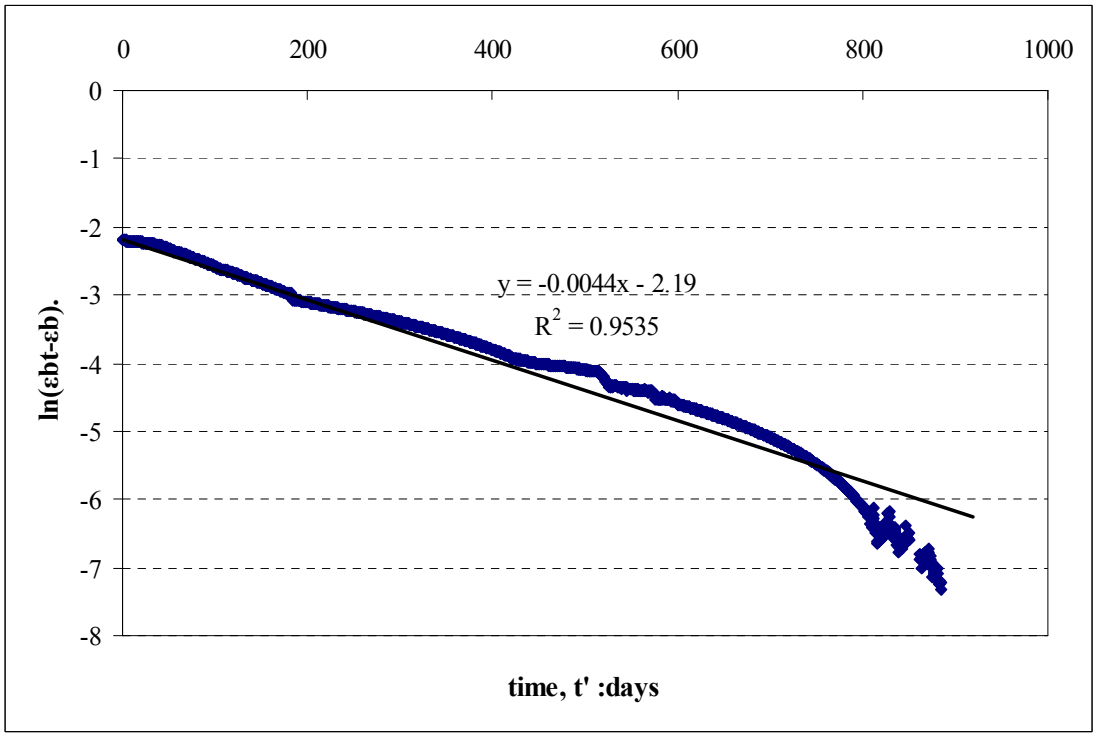


Figure 5.19: Determination of k_b for the raw MSW in CAR1 at 150 kPa

Table 5.10: k_b and ε_{bt} values for the pre-treated wastes and raw MSW

Waste	Degradation rate constant, k_b (day^{-1})	Total settlement strain due to biodegradation, ε_{bt} (%)	Gas generating potential (litre/kg DM)	Organic content, LOI (% DM)
UK MBT	0.0161	3.2	49.46	42.91
German MBT	0.0203	1.7	17.74	34.84
Raw MSW				
CAR1 (150 kPa)	0.0044	13.8	313.8	77.80
CAR2 (50 kPa)	0.0049	11.2	255.4	77.80

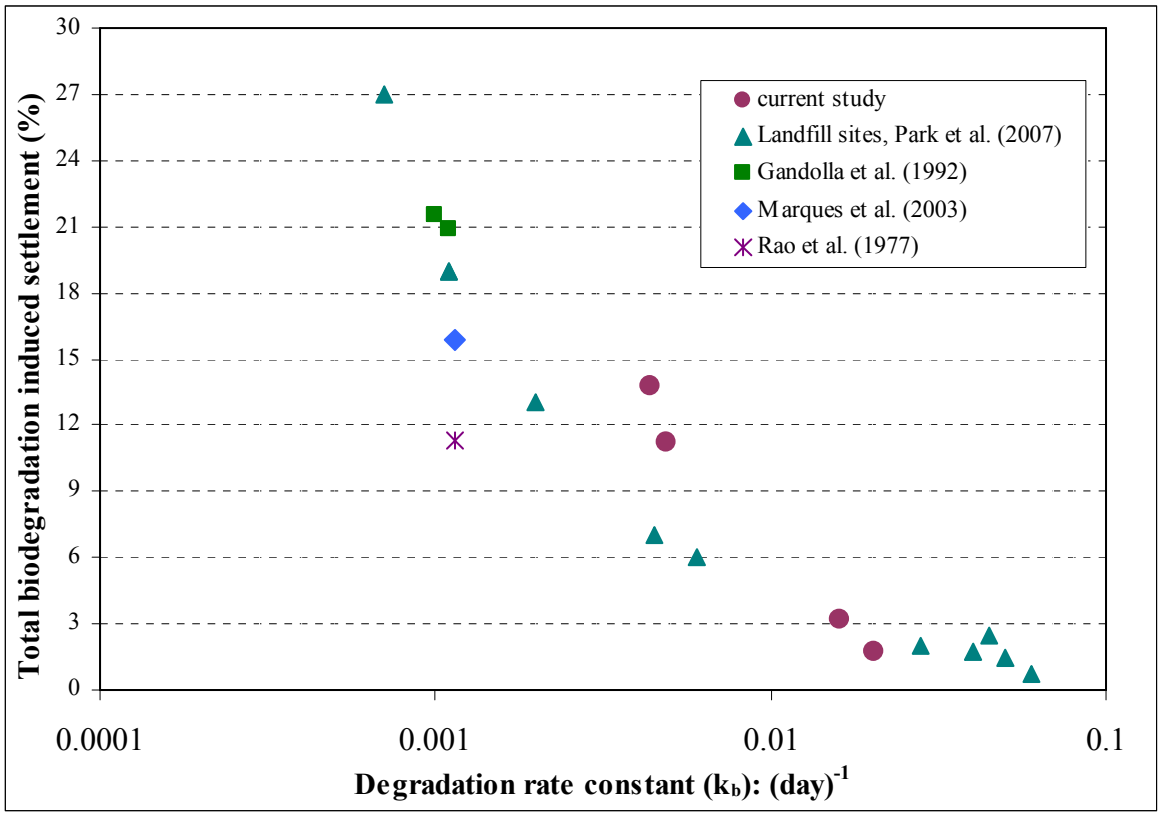


Figure 5.20: Relationship between k_b and ε_{bt}

Table 5.11: Published values of degradation rates k_b

Reference	Waste composition	Operational details	Degradation rate constant, k_b (day^{-1})	Total settlement strain due to biodegradation, ε_{bt} (%)
Marques et al. (2003)	56.3% organic, 12.4% paper & cardboard		9.51×10^{-4} to 1.10×10^{-3} Average: 1.14×10^{-3}	13.1-21.4 Average: 15.9
Gandolla et al. (1992)[a]	60% organic (dry)	waste shredded and mixed with sewage sludge	1.0×10^{-3} 1.1×10^{-3}	21.56 20.86
Rao et al. (1977) [a]	Nil		1.0×10^{-3}	11.33
Park et al. (2007)	MSW landfill sites		7.0×10^{-4} 1.1×10^{-3} 2.0×10^{-3} 4.5×10^{-3} 6.0×10^{-3} 4.5×10^{-2} 4.0×10^{-2} 5.0×10^{-2} 6.0×10^{-2}	27.0 19.1 13.5 7.2 6.1 2.6 1.7 1.5 0.7
Current study	UK MBT waste	Leachate recirculation, waste mixed with sewage sludge	1.61×10^{-2}	3.2
	German MBT waste	Leachate recirculation, waste mixed with sewage sludge	2.03×10^{-2}	1.7
	Raw MSW	Leachate recirculation, waste mixed with sewage sludge	4.4×10^{-3} 4.9×10^{-3}	13.8 11.2

Note: (1) [a] from Park and Lee (2002)

(2) Results of Park and Lee 2002 and Marques 2003 took out creep through their own models

The relationship between k_b and ε_{bt} for the data from the current study and previous studies (Table 5.11) is shown in Figure 5.20. There is a close inverse correlation between the degradation rate and the biodegradation-induced settlement. It is observed from the graph that high ε_{bt} values are associated with low values of k_b. In other words, wastes with higher organic content or higher gassing potential have low values of k_b (implying slower rates of degradation) than the wastes of low organic content or low gassing potential as shown for the current study in Table 5.10.

The values of k_b obtained in the current study are comparable with the field measurements (Park et al. 2007). Small differences may be due to the high moisture content, shredded state of the waste and the scale of the experiment in CARs ensuring homogeneous distribution of the moisture and constant favourable temperature conditions.

k_b values for pre-treated wastes of $1.6x10^{-2}$ day^{-1} (UK MBT waste) and $2x10^{-2}$ day^{-1} (German MBT waste) were higher than those found for raw MSW in CAR1 ($4.9x10^{-3}$ day^{-1}) and CAR2 ($4.4x10^{-3}$ day^{-1}). The values of k_b for the raw MSW were slightly higher but in the same order of magnitude as those reported by other researchers in Table 5.11.

Figs. 5.21 and 5.22 compare the observed and calculated biodegradation induced settlements for the UK and German MBT wastes. The observed and calculated results for the raw MSW in CAR1 (150 kPa) and CAR2 (50kPa) are compared in Figs. 5.23 and 5.24 respectively. The results show a close fit to the first order kinetic model for both pre-treated wastes and raw MSW. The model describes the experimental data fairly well, and successfully simulates the development of biodegradation induced settlement with time.

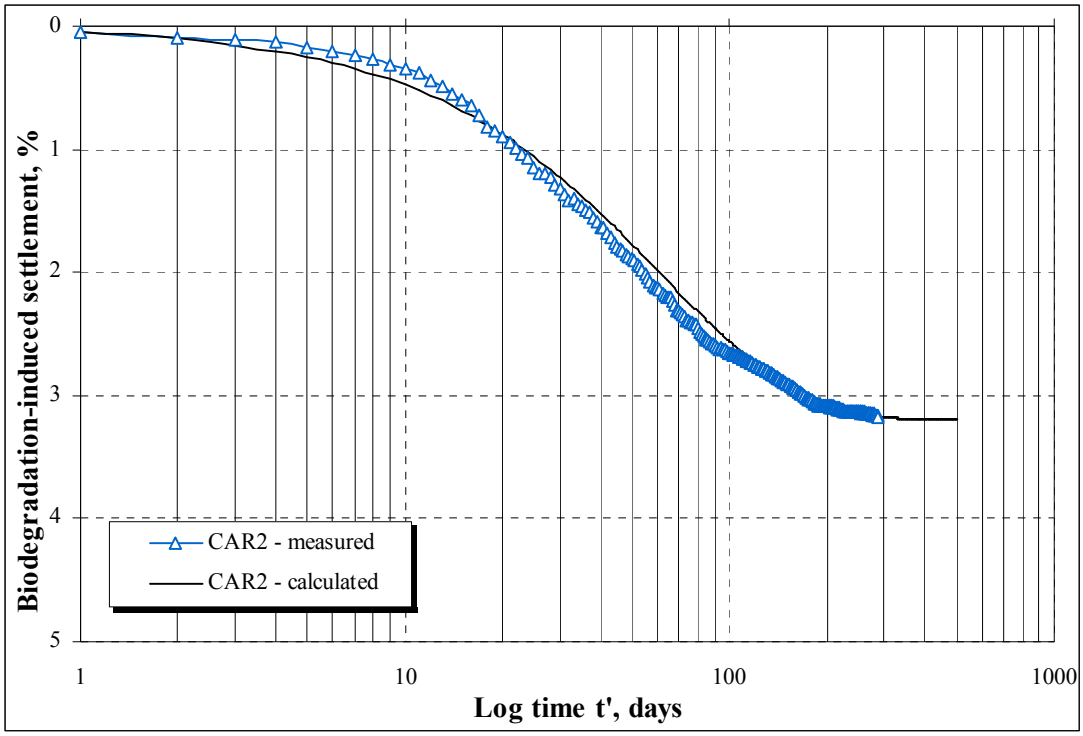


Figure 5.21: Actual and calculated biodegradation-induced settlements for the UK MBT waste

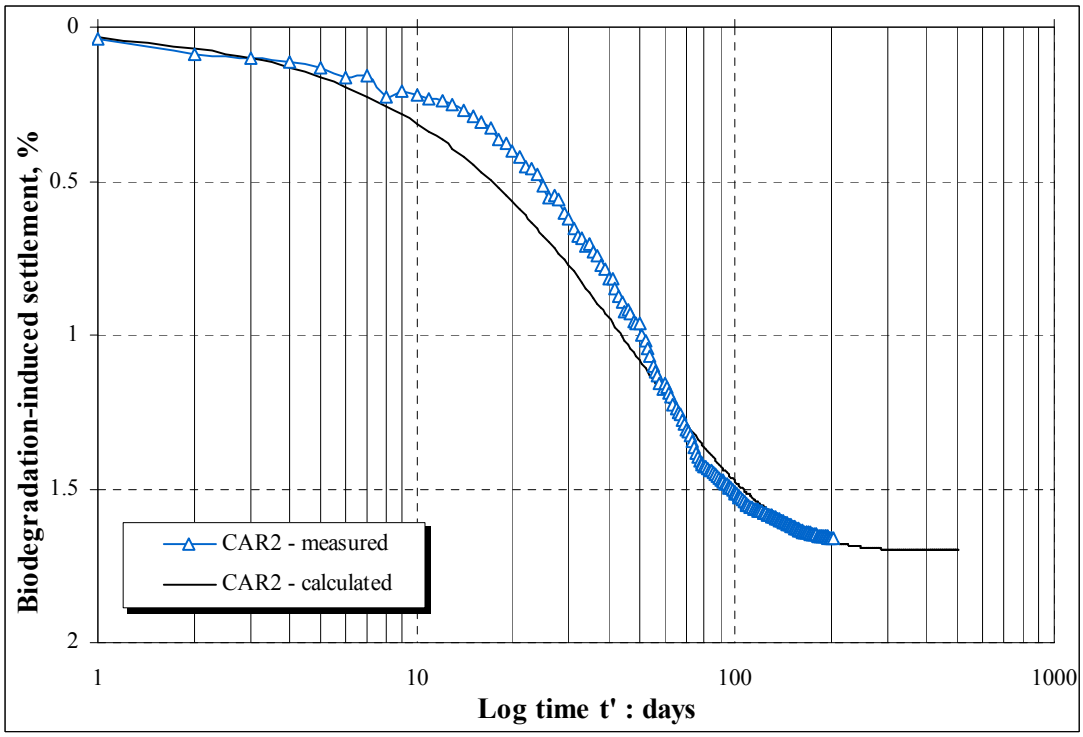


Figure 5.22: Actual and calculated biodegradation-induced settlements for the German MBT waste

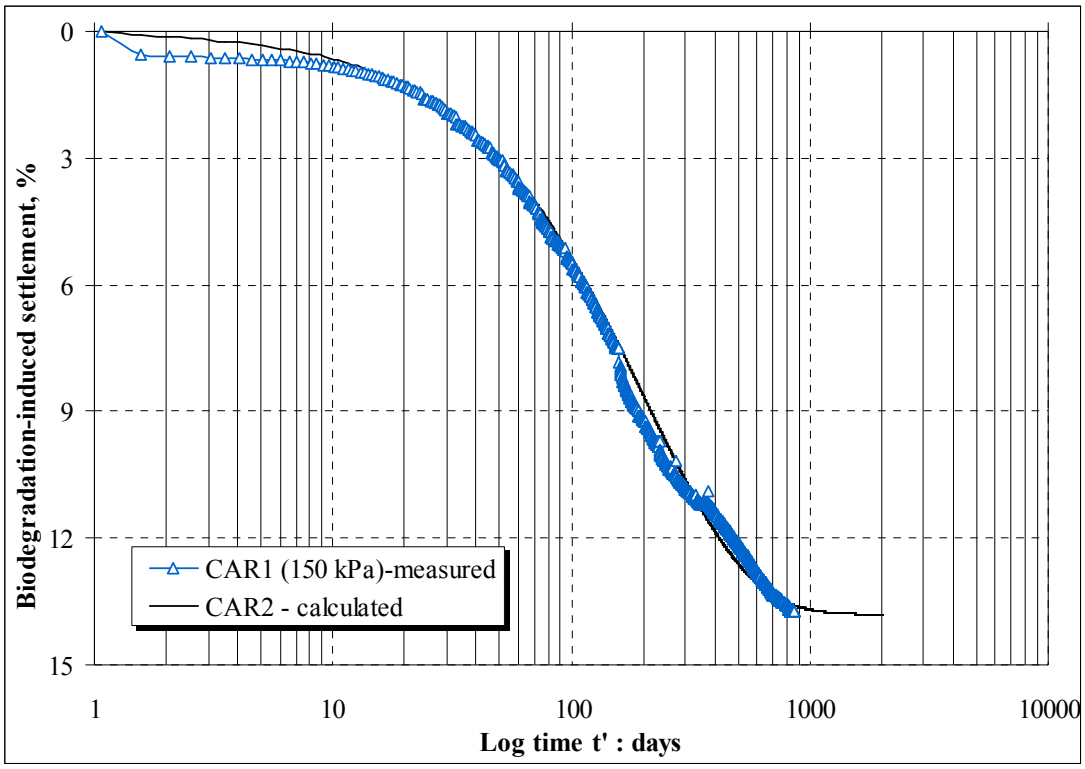


Figure 5.23: Actual and calculated biodegradation-induced settlements for raw MSW in CAR1 at 150 kPa

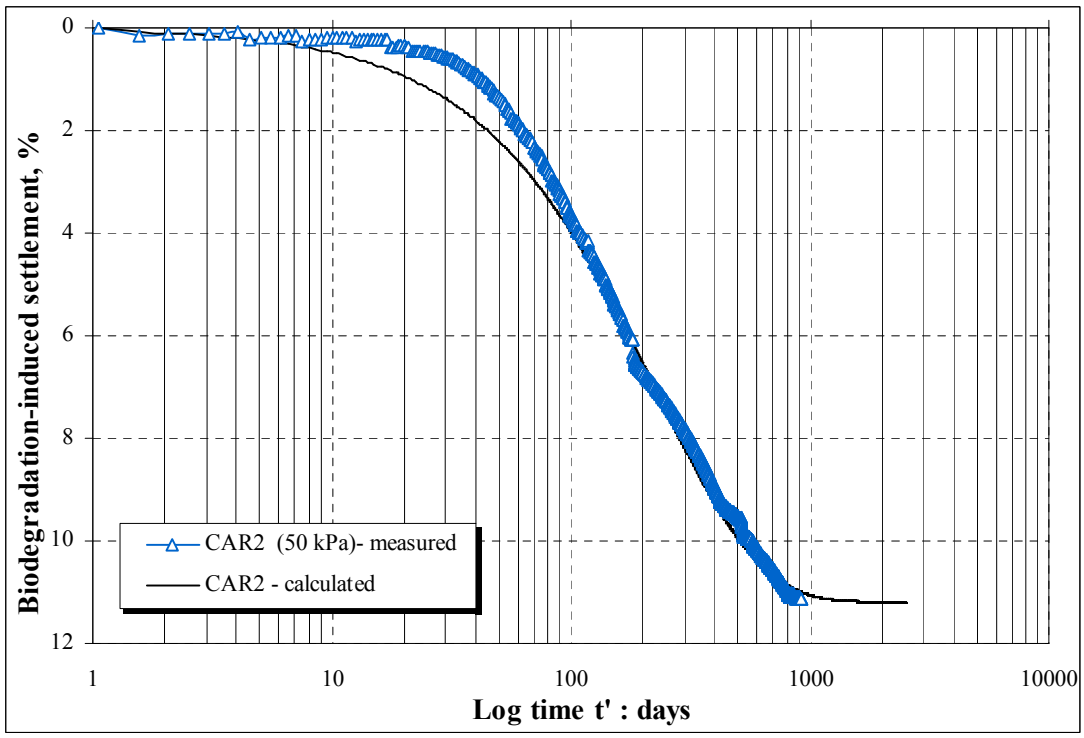


Figure 5.24: Actual and calculated biodegradation-induced settlements for raw MSW in CAR2 at 50 kPa

5.2.3 Total secondary settlement

Total secondary settlement results from both the mechanical creep and biodegradation, and may be expressed by summing equations 5.3 and 5.5 as

$$\varepsilon_{total} = \varepsilon_c + \varepsilon_b \qquad \text{Eq. 5.10}$$

$$\text{or} \quad \varepsilon_{total} = C_{\alpha\varepsilon} \log_{10}\left(\frac{t}{t_1}\right) + \varepsilon_{bt}\left(1 - e^{-k_b t'}\right) \qquad \text{Eq. 5.11}$$

The corresponding curves of total secondary settlement are compared to the observed curves in Figs. 5.25 and 5.26 for the UK and German MBT wastes respectively. Figs. 5.27 and 5.28 show the model approximations and the observed secondary settlement profiles for the raw MSW in CAR1 (150 kPa) and CAR2 (50 kPa) respectively.

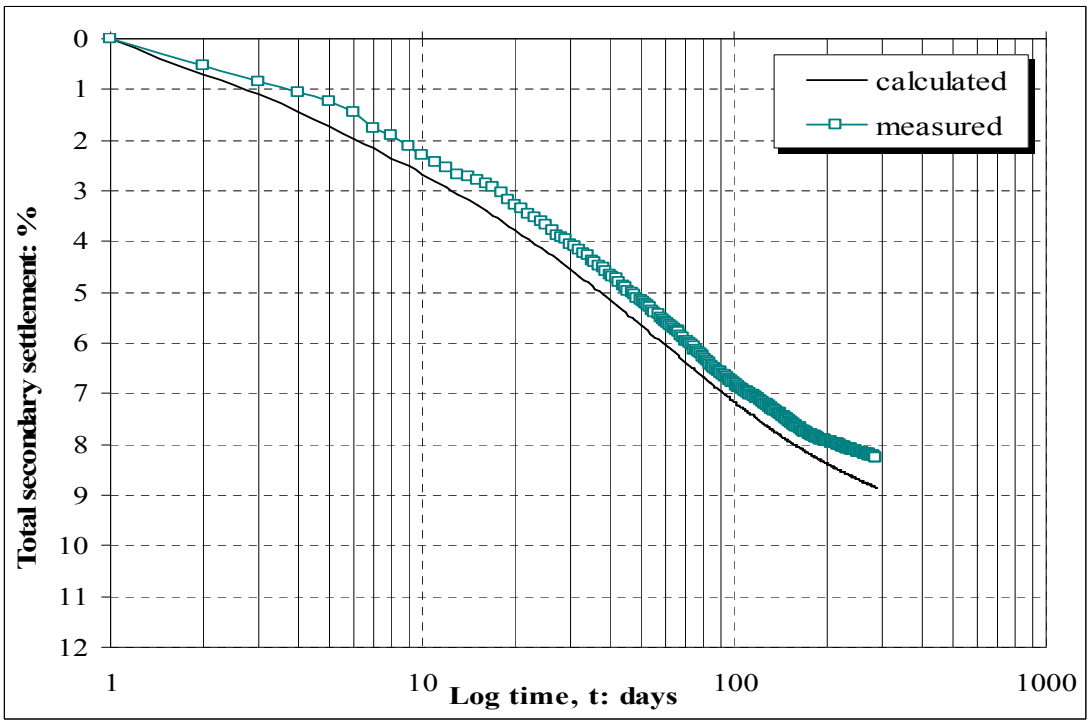


Figure 5.25: Measured and calculated total secondary settlement for the UK MBT waste

The model provides a good approximation to the settlement results. It might be possible to make a prediction of the evolution of long term settlement as a function of time using appropriate values of the model parameters. However, the model parameters might differ from site to site and may be difficult to estimate. Laboratory tests can furnish parameters to estimate long term settlements in landfills using simple models. However, for a landfill having a life of several decades and earning/costing a large amount of money, a laboratory test taking 200 days or so is probably a worthwhile investment.

The creep parameter ($C_{\alpha\varepsilon}$) and the ultimate settlement due to biodegradation (ε_{bt}) are directly transferable and applicable to the landfill sites to predict the settlement. However, the k_b values obtained in the current study may not be representative of what can be achieved on the site-scale. The k_b values for the sites may be different due to non-ideal wetting conditions, unfavourable temperature and unshredded waste, which may change the biodegradation kinetics and will consequently impact on the rate of degradation. Ideal conditions similar to those in the lab reactors are very difficult to achieve on a large scale. However, the values of parameter k_b obtained from the laboratory tests, are consistent with the field data provided by Park et al. (2007).

This model can provide an approximate estimate of the ultimate strain and the general trend of settlement during the design stage. To improve knowledge of the parameter k_b , a large data base would be necessary.

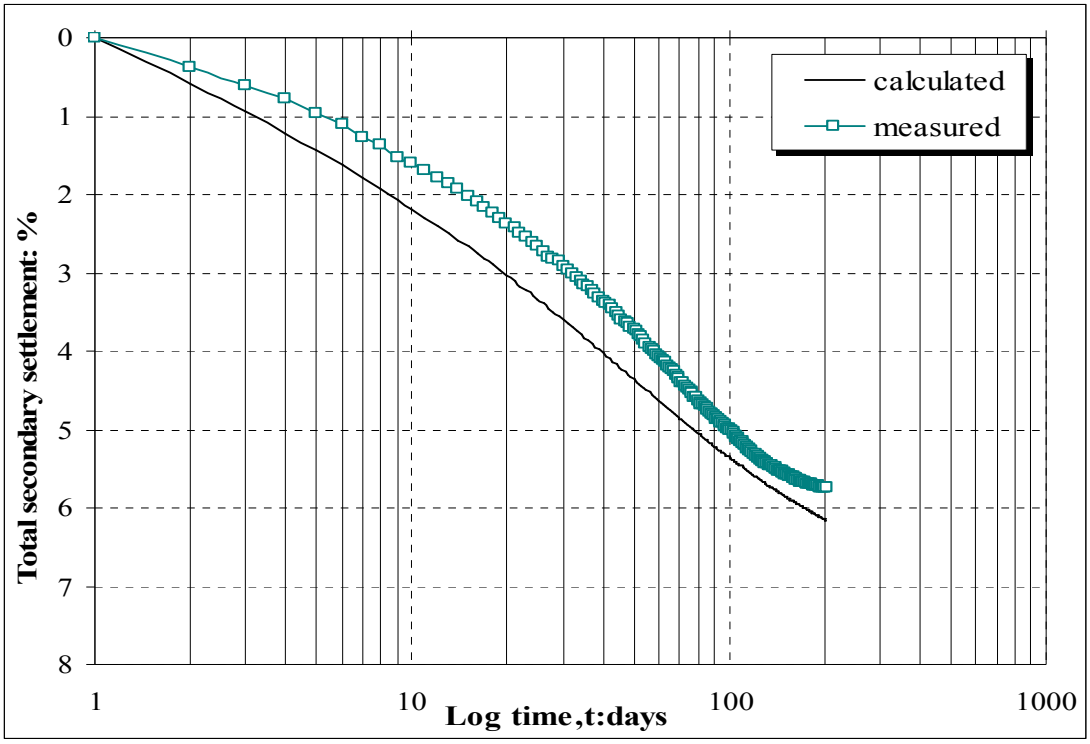


Figure 5.26: Measured and calculated total secondary settlement for the German MBT waste

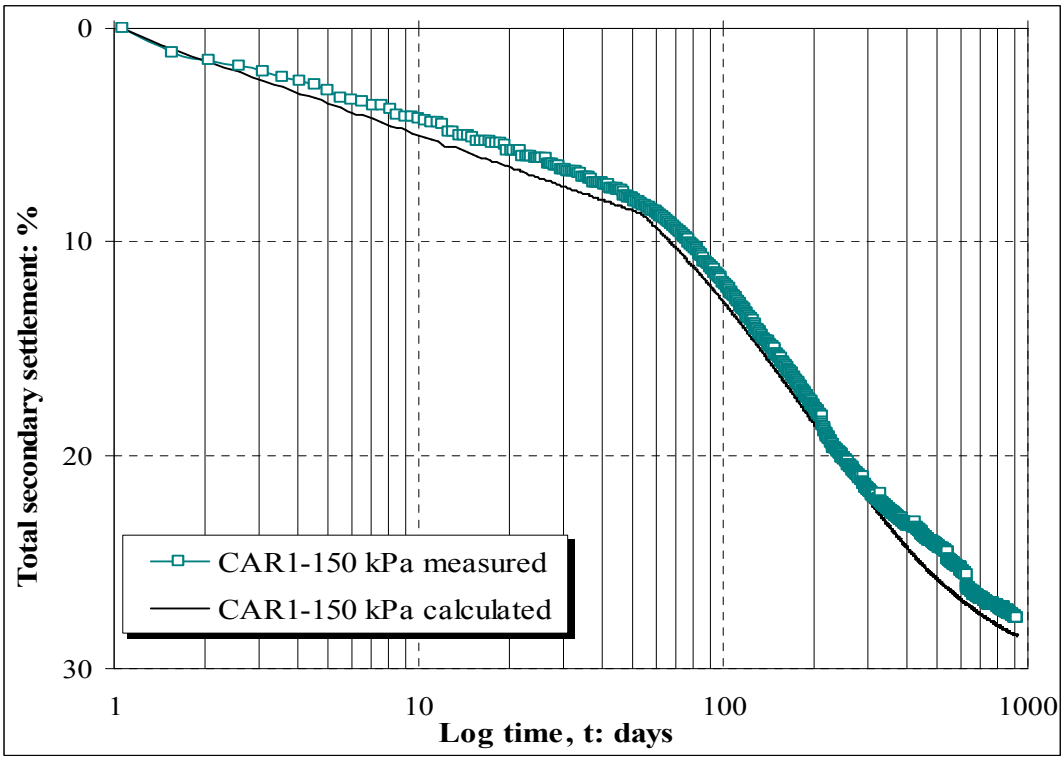


Figure 5.27: Measured and calculated total secondary settlement for raw MSW in CAR at 150 kPa

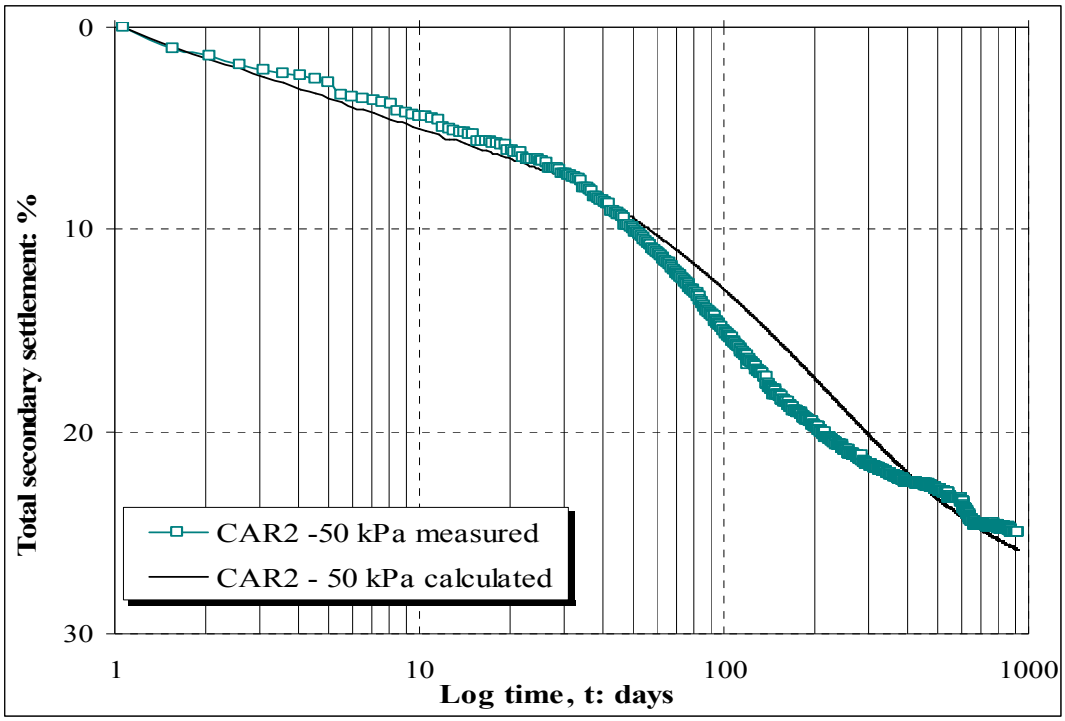


Figure 5.28: Measured and calculated total secondary settlement for raw MSW in CAR2 at 50 kPa

5.3 Hydraulic properties

The saturated hydraulic conductivity and drainable porosity of MBT waste were investigated at different applied stresses and densities. The procedure adopted to determine these parameters was described in Section 3.2.6. The waste in the CARs was subjected to two stages of compression, with applied stresses of 50 kPa and 150 kPa. The dry density of the wastes at the end of each compression stage is recorded in Tables 5.12 and 5.13 showing an increase from 0.44 t/m^3 to 0.71 t/m^3.

5.3.1 Saturated Hydraulic conductivity

The saturated hydraulic conductivity (k) of the UK and German MBT wastes was determined prior to compression (i.e. at zero applied load) and after

compression at applied stresses of 50 kPa and 150 kPa. The values of k for both wastes fall in the same order of magnitude as summarised in Tables 5.12 and 5.13. Over the stress range investigated, the saturated hydraulic conductivity reduced by 2 orders of magnitude, from approximately 7 x 10^{-5} m/s at zero load ($\rho_d \sim 0.44$ t/m^3) to about 5 x 10^{-7} m/s at an applied stress of 150 kPa (ρ_d~0.71 t/m^3). These values are comparable with those reported by Powrie and Beaven (1999), Reddy et al. (2009) and Bauer et al. (2006).

For the pre-treated wastes and raw MSW, the variation in saturated hydraulic conductivity with dry density, vertical stress and drainable porosity is presented in Figs 5.29, 5.30 and 5.31 respectively. The results demonstrate that the saturated hydraulic conductivity is controlled by the vertical stress through its impact on waste density and drainable porosity. The possible reasons for the decrease in k might be the reduction in pore size and change in geometry and continuity of pores resulting in reduction of fluid flow through the waste.

The decrease in saturated hydraulic conductivity with increasing dry density (Fig. 5.29) for the pre-treated wastes follows the same trend line as for raw MSW and the results are similar for the given range of dry density. The results of pre-treated wastes are consistent with the findings of Powrie & Beaven (1999). The results are also comparable to the range of 7.4x 10^{-5} to 4.6 x 10^{-6}m/s reported by Staub et al. (2009) for shredded MSW in a small range of dry density from 370 to 530 kg/m^3.

The trends of variation in hydraulic conductivity with vertical stress and drainable porosity are similar for the pre-treated wastes and raw MSW (Figs. 5.30 & 5.31). In both cases, the hydraulic conductivity of pre-treated wastes is

less than raw MSW by one order of magnitude. Figure 5.32 shows the increase in dry density of MBT waste in response to the increase in vertical stress. It shows that dry density of pre-treated wastes was much higher compared with raw MSW for a given vertical stress. Due to an increase in dry density resulting from the increase in applied stress, the drainable porosity decreases, which causes the saturated hydraulic conductivity to decrease.

Table 5.12: Summary of results for density, porosity and hydraulic conductivity of the UK MBT waste in CARs

Applied stress (kPa)	Dry density (Kg/ m^3)	Bulk Density (Kg/ m^3)	Drainable porosity (%)	Total porosity (%)	Flow (L/hr)	Hydraulic gradient (m/m)	Saturated hydraulic conductivity k (m/s)	Average k (m/s)
Control reactor (CAR1)								
Initial (no load)	442	679	41.02	65.17	5.64 8.82	0.107 0.180	8.12×10^{-5} 7.52×10^{-5}	7.82×10^{-5}
50 kPa	584	940	16.70	52.13	2.01 1.38	0.493 0.307	6.26×10^{-6} 6.9×10^{-6}	6.58×10^{-6}
150 kPa	680	1120	4.32	48.20	0.60 0.49	1.213 0.887	7.59×10^{-7} 8.48×10^{-7}	8.04×10^{-7}
Test reactor (CAR2)								
Initial (no load)	442	685	39.90	64.20	5.64 8.82	0.120 0.193	7.21×10^{-5} 7.01×10^{-5}	7.11×10^{-5}
50 kPa	582	949	15.80	52.89	2.01 1.38	0.507 0.367	6.09×10^{-6} 5.78×10^{-6}	5.94×10^{-6}
150 kPa	661	1091	4.48	47.50	0.60	1.320	6.97×10^{-7}	6.67×10^{-7}

Table 5.13: Summary of results for density, porosity and hydraulic conductivity of the German MBT waste in CARs

Applied stress (kPa)	Dry densit (Kg/ m^3)	Bulk Density (Kg/ m^3)	Drainable porosity (%)	Total porosity (%)	Flow (L/hr)	Hydraulic gradient (m/m)	Saturated hydraulic conductivity k (m/s)	Average k (m/s)
Control reactor (CAR1)								
Initial (no load)	442	672	37.77	60.69	5.64 8.82	0.127 0.213	6.84×10^{-5} 6.35×10^{-5}	6.59×10^{-5}
50 kPa	618	986	14.51	51.3	2.01 1.38	0.687 0.540	4.49×10^{-6} 3.92×10^{-6}	4.2×10^{-6}
150 kPa	706	1140	3.63	47.03	0.60 0.49	1.813 1.380	5.08×10^{-7} 5.45×10^{-7}	5.26×10^{-7}
Test reactor (CAR2)								
Initial (no load)	442	677	36.56	60.06	5.64 8.82	0.133 0.233	6.49×10^{-5} 5.80×10^{-5}	6.14×10^{-5}
50 kPa	614	984	14.13	51.5	2.01 1.38	0.740 0.573	4.17×10^{-6} 3.69×10^{-6}	3.93×10^{-6}
After degradation at 50 kPa for 202 days					2.01 1.38	1.100 0.827	2.80×10^{-6} 2.56×10^{-6}	2.68×10^{-6}
150 kPa	697	1145	3.82	46.74	0.60 0.49	2.080 1.787	4.43×10^{-7} 4.21×10^{-7}	4.32×10^{-7}

The German MBT waste had slightly lower k values than the UK MBT waste at a given applied stress. The reason could be the obstruction to fluid flow due to the presence of large plastic sheet fragments owing to the larger particle size of the German MBT waste.

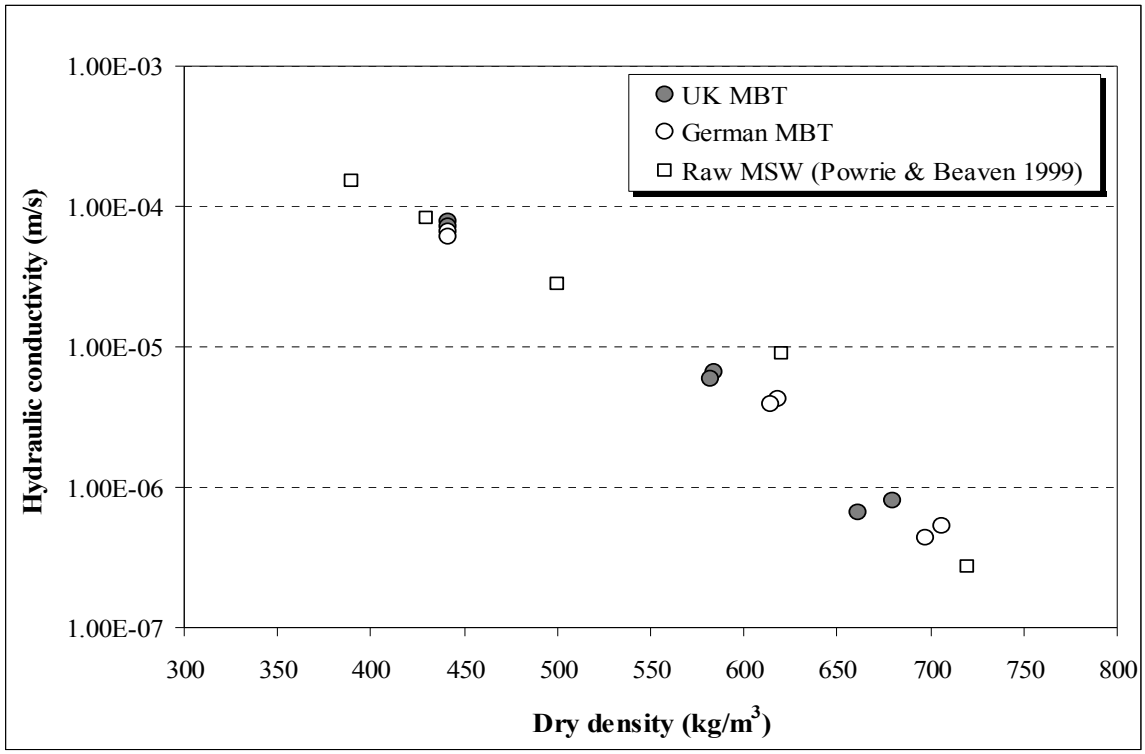


Figure 5.29: Variation of k with dry density for the pre-treated wastes and raw MSW

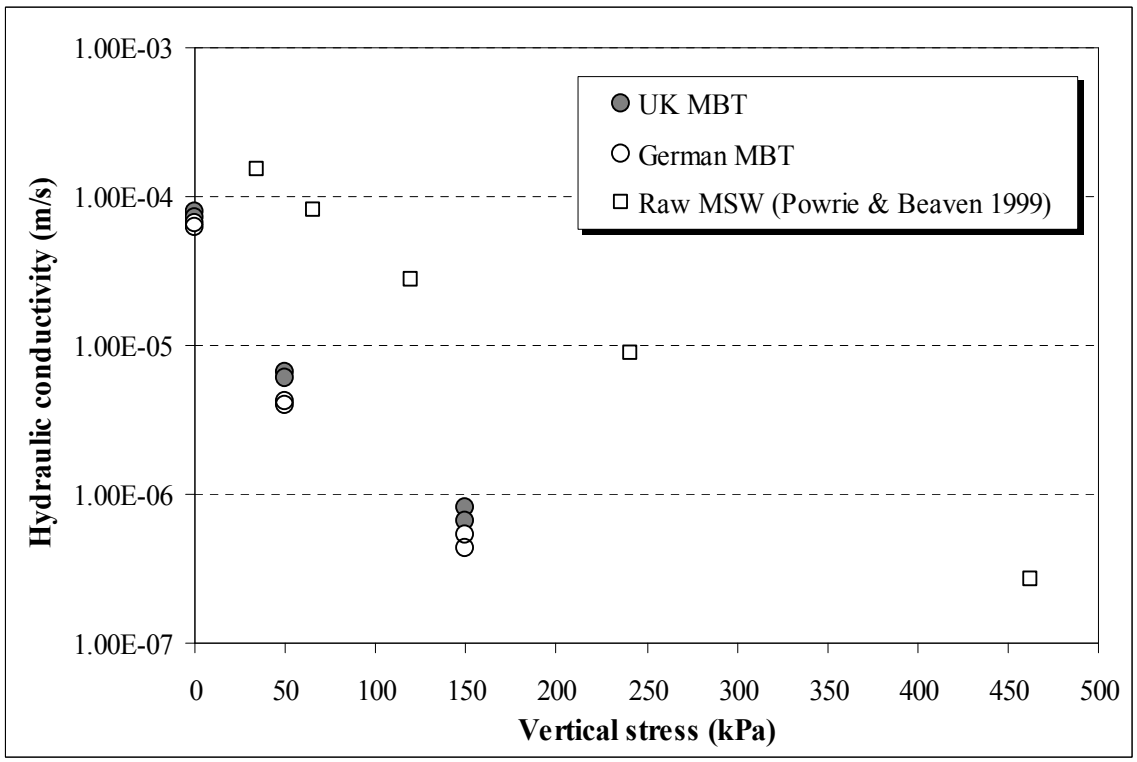


Figure 5.30: Variation of k with vertical stress for the pre-treated wastes and raw MSW

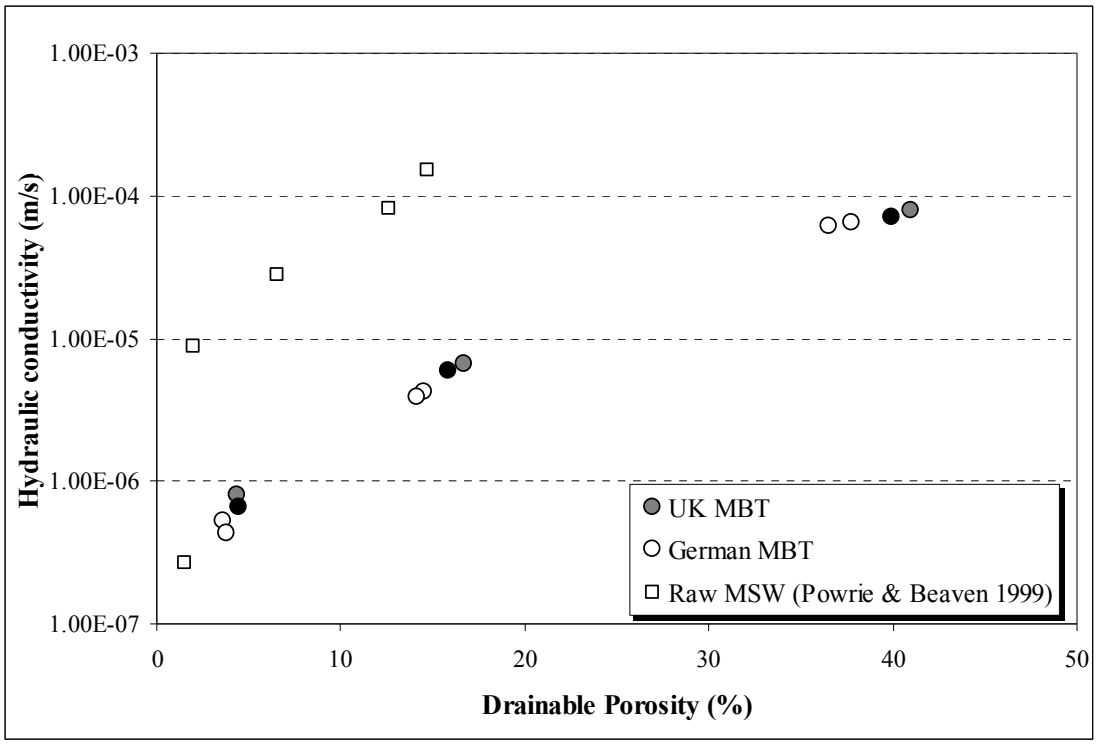


Figure 5.31: Variation of k with the drainable porosity for the pre-treated wastes and raw MSW

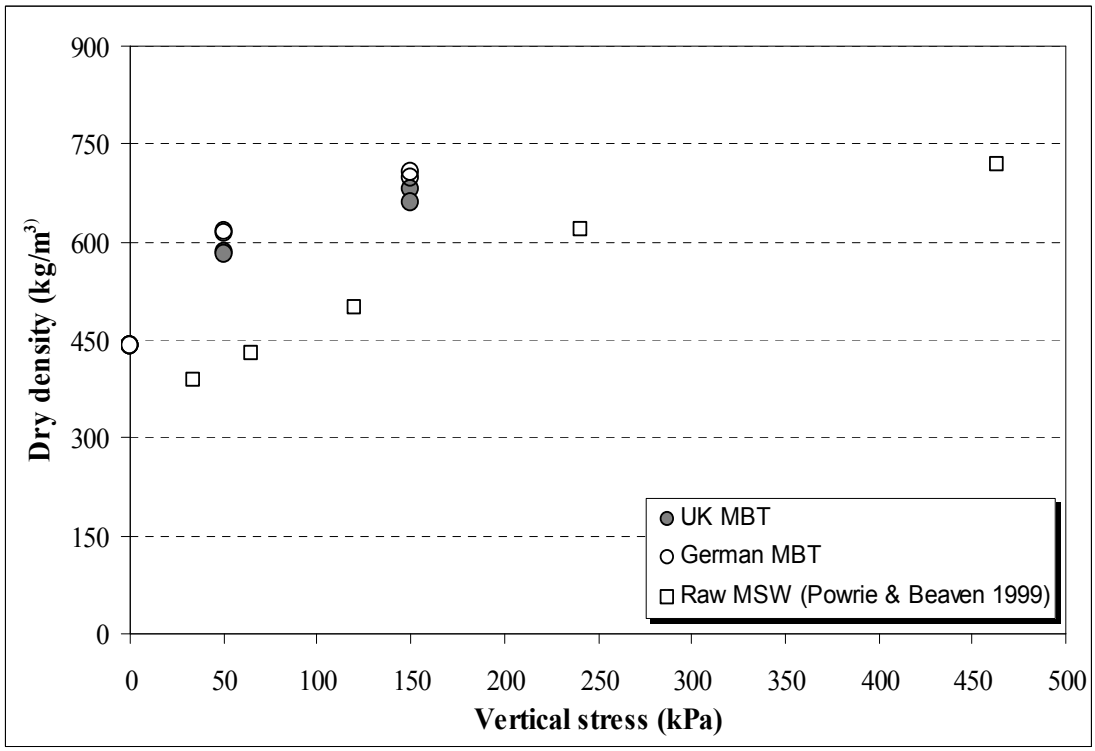


Figure 5.32: Variation of dry density with vertical stress for the pre-treated wastes and raw MSW

The saturated hydraulic conductivity of the degraded German MBT waste at an applied stress of 50 kPa was determined after 202 days of anaerobic degradation in CAR2 as shown in Table 5.13. The hydraulic conductivity changed slightly after degradation. This is an interesting observation which is consistent with Powrie & Beaven (1999) in that the differences in hydraulic conductivity resulting from particle size reduction and waste degradation are less significant. It is the compression and density that makes a significant change in k values.

5.3.2 Drainable porosity

As with the hydraulic conductivity tests, the drainable porosity of the UK and German MBT wastes was determined prior to compression (at zero applied stress) and after compression at applied stresses of 50 kPa and 150 kPa. Figures 5.33 and 5.34 show the piezometric head plotted against the volume of leachate drained from the waste. The gradient of the line through the data at any point is directly related to the drainable porosity as described in Beaven (2000). The drainable porosity results for the UK and German MBT wastes are shown in Tables 5.12 and 5.13 respectively.

The results indicate a reduction in drainable porosity at higher applied stress. Initially, both wastes had a relatively large drainable porosity of about 37% to 40% before compression i.e. at zero applied load. This was reduced to about 15-17% at an applied stress of 50 kPa, and to less than 5% at 150 kPa. The compaction of waste at higher stress tends to reduce the pore volume and the cross sectional area of flow paths causing a large decrease in drainable porosity.

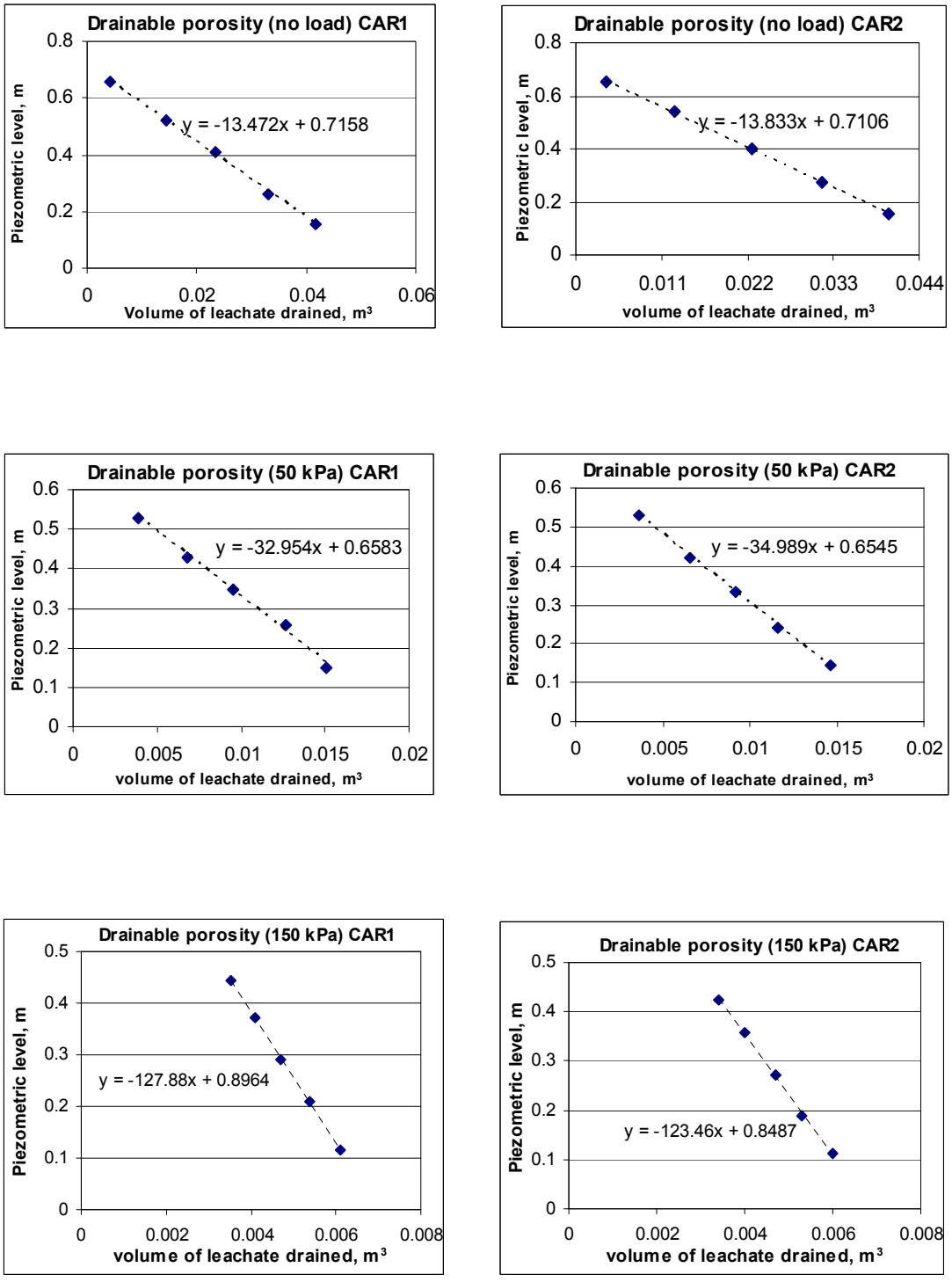


Figure 5.33: Drainable porosity determination of the UK MBT waste at applied stresses of zero load, 50 kPa and 150 kPa

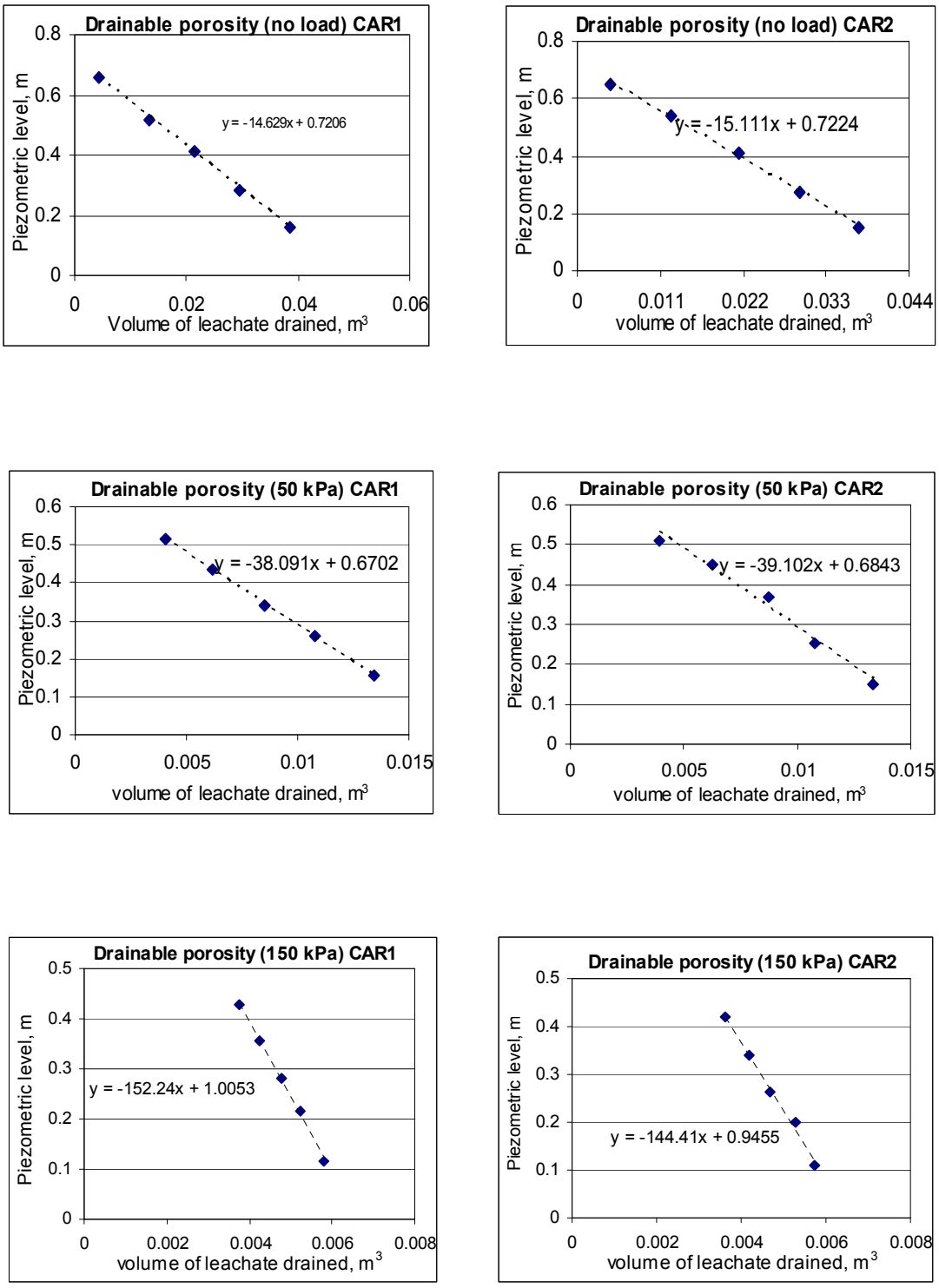


Figure 5.34: Drainable porosity determination of the German MBT waste at applied stresses of zero load, 50 kPa and 150 kPa

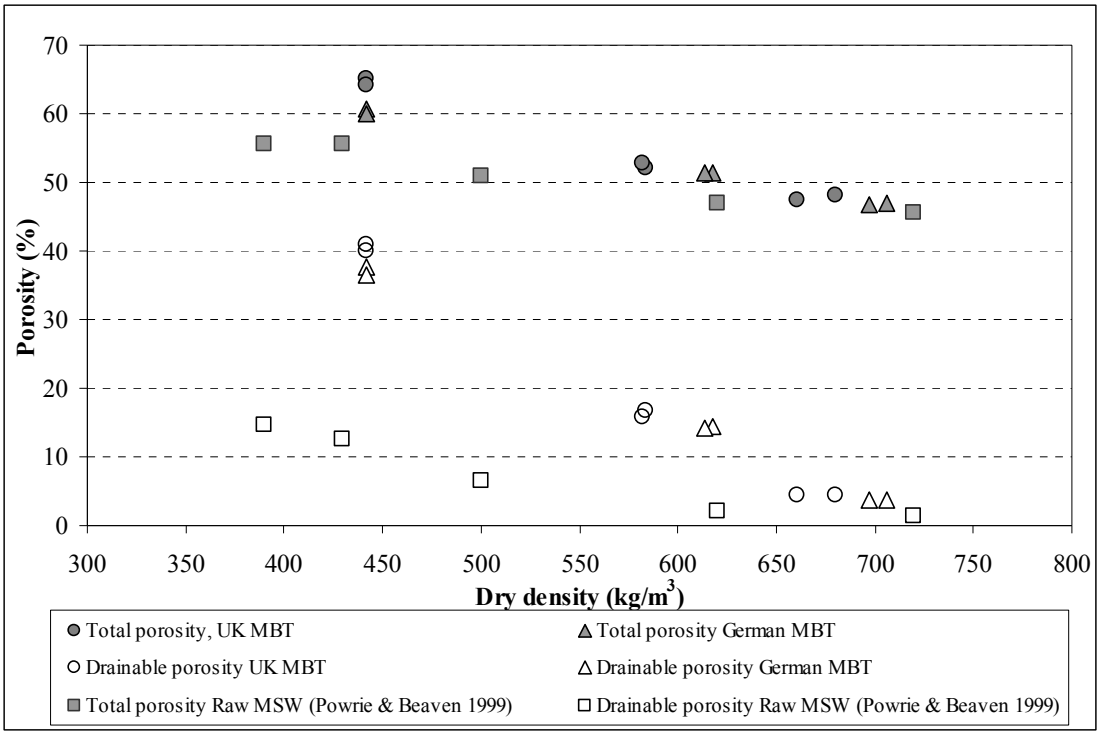


Figure 5.35: Total and drainable porosities of the pre-treated wastes and raw MSW

Figure 5.35 highlights a clear trend of decreasing total and drainable porosities with increasing dry density for both the pre-treated wastes and the raw MSW. The total porosities of raw MSW and pre-treated wastes are relatively similar with the pretreated waste samples being slightly more porous at lower density. There is little influence of an increase in dry density (resulting from increase in compression) on the total porosity. A change in waste composition and particle size of the pre-treated wastes had a relatively small influence on the total porosity.

For the pretreated wastes, the loss in drainable porosity on compression is greater than the loss in total porosity. The compression of pretreated wastes therefore has a greater influence, as seen from the loss of drainable porosity which is due to the reduction in drainable voids of the waste owing to

changes in waste structure. As dry density increases, the drainable porosity decreases which causes the vertical saturated hydraulic conductivity to decrease (Fig. 5.31). This behaviour is significantly influenced by the pore structure. Compaction tends to reduce the number and the cross sectional area of the flow paths, causing a large decrease of the drainable porosity.

In comparison to raw MSW, the drainable porosity of pre-treated wastes seems to display greater values at equivalent dry densities, and the values are comparable at higher density. Nonetheless, the loss of porosity on compression of raw MSW is entirely due to a loss of drainable porosity, i.e. there is a substantial reduction in the pore space available for liquid flow.

5.4 Degraded German MBT waste

After having degraded for 279 days, a sample of degraded German MBT waste of approximately 5 kg was retrieved from CAR2 for the analysis of the different waste components. The composition of the non degraded German MBT waste is compared with the degraded waste in Table 5.14.

Based on the composition of the degraded waste sample, it is observed that degradation results in an increase in the proportion of plastics and inert or inorganic components e.g. stones, ceramics, glass etc. However, the proportion and indeed the absolute amount of degradable constituents e.g wood and bones was decreased.

A major reduction was observed in the unidentified fraction consisting of particles greater than 5 mm whose material could not be identified as they are encased in soil-like material. It could be inferred that a part of unidentified fraction has degraded and produced gas.

Table 5.14: German MBT waste components before and after degradation

Component	Dry mass of non degraded German MBT (before emplacement in CAR2)		Dry mass of degraded German MBT (retrieved form CAR2 after 279 days degradation)	
	%	kg	%	kg
Flexible Plastics	2.4	0.96	3.14	1.222
Rigid Plastics	5.91	2.364	9.43	3.669
Wood	3.22	1.288	1.96	0.762
Textile	0.63	0.252	0.64	0.249
Rubber	0.25	0.1	0.13	0.050
Bones	0.37	0.148	0.09	0.035
Metal	1.49	0.596	0.52	0.202
Ceramics	4.25	1.7	6.65	2.588
Stones	3.17	1.268	3.89	1.514
Glass	24.36	9.744	25.33	9.856
Paper	0.18	0.072	0.49	0.190
Unidentified < 5mm	27.02	10.808	33.97	13.218
Unidentified > 5mm	26.75	10.7	13.75	5.350
Total	100.00	40	100.00	38.91

Degraded German MBT waste gradation was carried out using sieve analysis in accordance with BS1377-2 (1990). The particle size distribution curves of the degraded and non-degraded German MBT waste are compared in Figure 5.36. The difference between the curves is not significant except that the degraded waste contained more fines. The proportion of "unidentified <5mm" fraction in the waste had increased from an initial value of 27% to about 34% after degradation as shown in Table 5.14. This fraction represents a mixture of different components less than 5 mm that could not be identified or further separated owing to the small particle size. Particle disintegration

or breakdown with decomposition may have resulted in an increase in the proportion of finer particles.

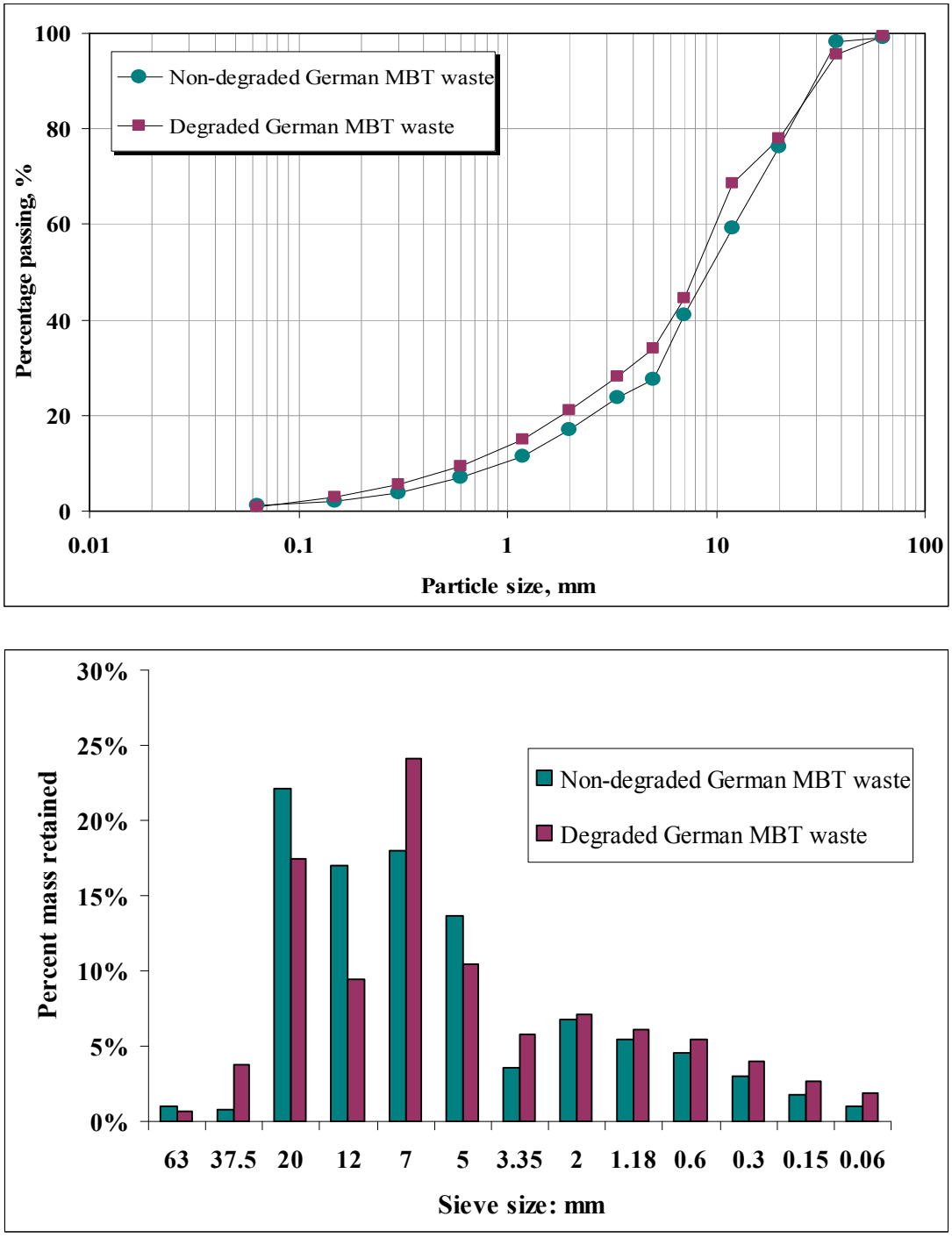


Figure 5.36: Particle size distribution curve for the German MBT waste before and after degradation

CHAPTER 6 SUMMARY AND CONCLUSIONS

This chapter presents the summary and key conclusions drawn from the work and outlines areas of future research.

Comprehensive laboratory scale experiments were carried out to investigate the long term biodegradation and settlement behaviour of MBT wastes treated to typical UK and German standards under simulated anaerobic landfill conditions. Large scale CARs were set up to evaluate the gas generating potential, leachate quality, settlement characteristics and hydraulic properties of MBT wastes. Small scale BMP reactors were employed to characterise the anaerobic biodegradability of MBT waste in terms of the change in solids composition with the progression of decomposition. The work presented has enhanced fundamental knowledge of the biodegradation and settlement behaviour of MBT waste, and of the potential benefits by comparing the pre-treated wastes with raw MSW. A detailed characterisation of the waste and its associated chemical and physical properties was a key component of the study. The gas generation potential, leachate quality and settlements have demonstrated that waste stabilisation was achieved in less than a year under enhanced biodegradation conditions in the CARs for the UK and German MBT wastes.

6.1 Summary

The findings from the previous chapters are summarised below.

6.1.1 Biogas

The total volume of biogas produced at STP for the UK and German MBT wastes was 49.46 litres/kg DM and 17.74 litres/kg DM respectively. The higher gassing potential of the UK MBT waste could be explained by the lower degree of biological pretreatment, evidenced by the higher values of organic content compared with the German MBT waste. Most of the biogas was produced in the first 200 days for the UK MBT waste and in the first 150 days for the German MBT waste. Gassing then continued at a much lower rate until day 280 and 195 when it had effectively ceased for the UK and German MBT wastes respectively. Though the rate of biogas production decreased towards the end, the methane and carbon dioxide gas concentrations remained unchanged at about 60% and 35% respectively. The cumulative gas volume produced by degradation of raw MSW was significantly higher (255.4 litres/kg DM) and the gassing rate was also much higher, in the range 2 - 8 litre/kg DM/day compared with 0.4 - 0.8 litre/kg DM/day range in the pre-treated wastes. About an 80% reduction in the gas generating potential was achieved after 6 weeks of pretreatment (UK MBT waste), whilst prolonging the duration of pretreatment up to 9 weeks (German MBT waste) gave an overall reduction of about 92%.

The lower rates of gas production may be problematic for effective gas collection and management systems, especially for energy recovery systems. Further work is needed to investigate if it is an economically viable option.

The results showed no evidence of an acidogenic phase in the degradation of the pre-treated wastes: in contrast, raw MSW (Ivanova et al. 2008*a*) experienced a long acidogenic phase of about 55 and 32 days in CAR1 (150

kPa) and CAR2 (50 kPa) respectively. The addition of acids to the control reactor was successful in inhibiting biodegradation over the entire duration of the tests.

6.1.2 Leachate

This study has measured the evolution of the chemistry of the leachate from MBT wastes and has significantly improved the knowledge of leachate in MBT waste landfills.

The organic strength of the leachate from the German MBT waste was low compared with that from the UK MBT waste owing to the different biological processing steps during the pretreatment. Raw MSW gives a high organic load in the leachate which may be attributed to higher LOI and TC contents. TOC load in leachate of raw MSW was about 2.1 g/kg DM compared with 1.1 g/kg DM in the UK MBT waste and 0.58 g/kg in the German MBT waste.

The leachate load of ammoniacal nitrogen from the German MBT waste was less than that from the UK MBT waste which was probably due to the lower nitrogen content of the German MBT waste. Raw MSW generate a higher leachate load of ammoniacal nitrogen ~ 3.2 g/kg DM compared to the UK MBT waste (~0.8 g/kg DM) and German MBT waste (~0.5 g/kg DM).

A significant benefit of waste pretreatment is evident from the reduced level of ammoniacal nitrogen and TOC contents in the leachate, which are substantially lower for the pre-treated wastes than the raw MSW. Due to the reduced landfill pollution potential, the timescales over which leachate will require management can be reduced significantly.

Heavy metals concentrations were lower in the leachate from the German MBT waste than in that from the UK MBT waste. Immobilisation of heavy metals was completed within the first 3-4 weeks due to the establishment of highly reducing environment. Low concentrations of heavy metals in the leachate primarily reflect the low amount of metals in the waste and their low solubility. Leaching of heavy metals from raw MSW was more than from pre-treated wastes. The pretreatment may have either reduced the metal content in the waste or make the metals less mobile which is a positive effect.

6.1.3 Settlement

The majority of settlement occurred as immediate compression in response to the loading, accounting for a settlement of ~19% and ~22% of the initial waste height in the UK and German MBT wastes respectively. A total time of 24 hours was shown to be the time required for completion of primary settlement for the MBT waste samples tested in CARs. Primary (consolidation) settlement resulted in additional settlements of 5.9% to 7.8% of the waste height after immediate settlement in the UK and German MBT wastes.

With the available data, the contribution of mechanical creep and biodegradation to settlement could be quantified separately and compared as the control reactor (CAR1) remained inhibited over the entire duration of the tests. The long term secondary settlement accounted for further settlements of 8.26% and 5.75% of the waste height after primary settlement for the UK and German MBT wastes respectively. It was concluded that the settlement caused by mechanical creep was 5.09% and 4.09%, and the settlement associated with biodegradation was 3.17% and 1.66% for the UK and German

MBT wastes respectively. Secondary settlement due to mechanical creep was more significant than that due to biodegradation, highlighting the importance of mechanical creep on the overall magnitude of secondary settlement.

The long term secondary settlement of pre-treated wastes was much smaller than for raw MSW, due mainly to the high amount of biodegradation induced settlements. Results confirmed the strong relationship between biodegradation induced settlement and the cumulative biogas production.

Based on the two distinct settlement mechanisms of creep and biodegradation, simplified models were used as a mathematical tool for the evaluation of the long term settlement of pre-treated wastes and raw MSW. The parameters of the model, $C_{\alpha\varepsilon}$ and k_b, were evaluated from the settlement data and the model calculations were compared with the experimental results. $C_{\alpha\varepsilon}$ values for the pre-treated wastes were found to lie within a narrow range, and creep of the waste was slightly affected by the stress but probably the governing factor was density.

The kinetic model successfully simulates the form of the biodegradation induced settlement. Results showed close agreement between the simulated and experimental total secondary settlements in the CARs for the UK MBT waste, German MBT waste and raw MSW. It suggests that laboratory tests can furnish parameters to estimate long term settlement using simple models.

6.1.4 Anaerobic biodegradability

Degradability of MBT waste was evaluated using LOI, TC, biogas potential, cellulose, hemicellulose and lignin contents of decomposing waste in small scale BMP reactors. A strong correlation was found between the biogas potential and the (C+H)/L ratio. The initial cellulose and hemicellulose contents of the pre-treated wastes were reduced, but in the degraded state were very similar to degraded MSW indicating that the same state of final decomposition is achieved. LOI content of the waste samples decreased with the progression of degradation which is consistent with the depletion of the TC content. The BMP test results indicate that changes in LOI, TC and (C+H)/L ratio and biogas potential are inter-related.

6.1.5 Hydraulic properties

Data from both MBT wastes followed the same general trend of decreasing hydraulic conductivity and drainable porosity with increasing stress and waste density. The drainable porosity was reduced to about 15-17%, at an applied stress of 50 kPa and to less than 5% at 150 kPa stress. Not surprisingly, the density had a major influence on the drainable porosity values. The saturated hydraulic conductivity of MBT waste samples was in the range $4x10^{-6}$ to $6x10^{-6}$ m/s at 50 kPa, reducing to $5x10^{-7}$ to $8x10^{-7}$ at 150 kPa. The hydraulic conductivity of pre-treated wastes and raw MSW are similar for the given range of dry density and the results from this study on pre-treated wastes are consistent with the findings on raw MSW by Powrie and Beaven (1999). The hydraulic conductivity of German MBT waste was slightly affected after degradation at 50 kPa.

6.1.6 Mass balance

The carbon mass balance accounted for about 95 – 98% of carbon initially in the waste. The measured nitrogen content at the end of the experiment accounted for about 94 – 96% recovery of the initial nitrogen. The carbon and nitrogen mass balances indicate only a small proportion of the carbon and nitrogen in the system is unaccounted for. A large proportion of carbon and nitrogen remain locked up in the waste material and is not released. Despite the lack of closure of the mass balance, the calculations provide very encouraging results.

6.2 Conclusions

The following conclusions have been drawn from this study:

1. The settlement of the waste can quite easily be divided into immediate compression, primary settlement and secondary settlement. At the laboratory scale, the primary settlement (consolidation) has been shown to be completed in 24 hours. The contributions of mechanical creep and biodegradation to secondary settlement can be identified and quantified separately.
2. The long term settlement curve for the pre-treated wastes and raw MSW can be reproduced with a model based on a logarithmic law for creep and a kinetic model for the biodegradation induced settlement. Primary settlement of a saturated waste can be assessed using Terzaghi's theory of one dimensional consolidation.
3. Laboratory tests can furnish parameters to estimate long term settlements in landfills, using simple models. The parameters might

however differ from site to site and may be difficult to estimate. In reality, for a landfill having a life of several decades and earning/costing a large amount of money, a laboratory test for a period of 200 days or so is probably not a major problem.

4. Irrespective of whether it is MBT waste or raw MSW, the relationships for hydraulic conductivity against dry density, biodegradation induced settlement against gas produced, and change in cellulose plus hemicellulose ratio versus biogas potential are all similar, with pre-treated wastes being generally at one end of the scale.
5. The pretreatment of raw MSW substantially reduces the gas generating potential but the lower rates of gas production may be problematic for effective gas collection and management systems.
6. A significant benefit of waste pretreatment is evident from the reduced level of TOC, ammoniacal nitrogen and heavy metals contents in the leachate, giving a reduced potential for pollution.
7. The long term secondary settlement of pre-treated wastes was much smaller than for raw MSW.

6.3 Future work

The following suggestions for future work can be made:

1. Further experiments are required to investigate the nitrogen removal mechanism in CARs, which may be due to several reasons e.g. anammox process, sorption, precipitation as struvite.
2. The unidentified fraction of the UK and German MBT waste accounted for more than 50% by mass. This fraction was found to

decrease after degradation and therefore need to be explored in further studies.

3. The pretreatment of raw MSW may have either reduced the metal content in the waste or make the metals less mobile which requires further investigations.
4. To have complete knowledge of the leachate emanating from MBT waste landfills, it is recommended to investigate the presence of potentially harmful trace organic substances in leachate e.g. xenobiotic organic compounds (XOCs) consisting of hydrocarbons, pesticides, aromatic and aliphatic organic substances and, organo-metal compounds e.g. di-methyl mercury, tetra-methyl lead etc.

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