Innovative landfill bioreactor systems for municipal solid waste treatment in East Africa aimed at optimal energy recovery and minimal greenhouse gas emissions

Fredrick Mathew Salukele

Thesis committee

Promoter
Prof. dr. ir. W.H. Rulkens
Professor of Environmental Technology
Wageningen University

Co-Promoter
Dr. ir. J.C.L. van Buuren
Assistant Professor, Sub-department of Environmental Technology
Wageningen University

Other members
Prof. dr. ir. G. Spaargaren, Wageningen University
Prof. dr. hab. inż. G. Malina, AGH University of Science and Technology, Krakow, Poland
Dr. S.M. Mgana, Ardhi University, Dar es Salaam, Tanzania
Dr. ir. A.H.M. Veeken, Attero B.V., Haelen, Netherlands

This research was conducted under the auspices of the Graduate School for Socio-Economic and Natural Sciences of the Environment (SENSE)

Innovative landfill bioreactor systems for municipal solid waste treatment in East Africa aimed at optimal energy recovery and minimal greenhouse gas emissions

Fredrick Mathew Salukele

Thesis
submitted in fulfillment of the requirements for the degree of doctor
at Wageningen University
by the authority of the Rector Magnificus
Prof. dr. M.J. Kropff,
in the presence of the
Thesis Committee appointed by the Academic Board
to be defended in public
on Wednesday 4 September 2013
at 1.30 p.m. in the Aula.

Fredrick Mathew Salukele
Innovative landfill bioreactor systems for municipal solid waste treatment in East Africa aimed at optimal energy recovery and minimal greenhouse gas emissions
204 pages

PhD thesis, Wageningen University, Wageningen, NL (2013)
With references, with summaries in English and Dutch
ISBN 978-94-6173-633-8

Acknowledgements

The journey was not simple at all, there were ups and downs, but I finally made it to the end. Indeed this is one of the most memorable projects that I have ever undertaken. I certainly did not accomplish this thesis alone, many individuals and organizations contributed in different ways and at different stages. So, I hereby acknowledge those who are not mentioned on the cover but without their support, this work would have not been completed.

I sincerely express my deepest gratitude to my promoter and co-promoter Prof.dr.ir Wim H. Rulkens and Dr.ir. Joost van Buuren respectively for providing overall guidance for this study. Wim, your critical insights and broad perspective motivated the scientific quality of this thesis. Your excellent guidance, support and comments while in the Netherlands and when I was in Tanzania are highly appreciated. Sincere thanks are also due to your wife Riet Rulkens who was always very friendly, encouraging and supportive whenever I met her during the various dinners we had together. Joost, whom you always called me "chief," I am grateful for your field visits during different stages of the research work in Mwanza and Dar es Salaam. Your open and constructive feedback provided insights that helped me to carry out the research and the writing of the thesis. Thank you both for the time you took to go through the countless drafts and the contributions you made to all chapters. I have learnt a lot in this PhD which I believe it is the process that I have gone through that has shaped my thinking and made me more analytical today.

My sincere gratitude goes to INREF - Wageningen University for the financial support offered to me through the PROVIDE project. I wish to extend my deep felt thanks to Dr. Gábor Szántó the PROVIDE coordinator and a very good friend, for his endless encouragement and tireless efforts to make my entire PhD programme run smoothly both in Tanzania and in the Netherlands. Corry Rothuizen, thank you for handling the administrative and logistics ever since before arriving in Wageningen until today, as I always said, you are the best. Thanks to all PROVIDE administrators, Dr.ir. Peter Oosterveer, Prof.dr.ir. Gert Spaargaren, Prof.dr.ir. Arthur Mol, Christa de Bruin (who later left the project) and all others in the running of the project.

To the PROVIDE PhD team Judith Tukahirwa, Sammy Letema, Thobias Bigambo, Mesharch Katusiimeh, Richard Oyoo, Aisa Solomon, Christine Majale, Kenyanito Toure, and Maurice Onyango, I am so glad I met and to have you as my colleagues, friends, brothers and sisters. Without you, this PhD would not be what it is now.

I also appreciate the advice and support I got from Prof.dr.ir Jules van Lier from UNESCO-IHE whom I started with in this journey of PhD. I am also indebted to express my gratitude to the East African PROVIDE project partners Dr. James Okot-Okumu from Makerere University (Uganda), Dr. Caleb Mireri from Kenyatta University (Kenya) and especially to Dr. Shaaban Mgana from Ardhi University (Tanzania) who initially recommended me to the PROVIDE project and continued to support and assist me to shape the research and thesis. I am also grateful to my employer Ardhi University for granting me a study leave, space for construction of my Pilot scale reactors and the use of the Environmental Engineering laboratory for various analyzes. Many thanks to Ramadhani Mbulume from Ardhi University for his support in the laboratory and operation of the pilot scale Landfill bioreactors.

I want to thank all those working at the Sub-department of Environmental Technology at the Biotechnion and later at Technotron. Special thanks to Liesbeth Kesaulya-Monster for making my working environment in the department very pleasant and for handling lots of administrative matters related to my study. Thanks to Gusta, Romana and Anita and all of those whom we shared an office at different times (Kim Oanh, Zhang Lei, Mieke, Kanjana, Diego, Simon, Wei-Shan (Momo), Zhengyu (Scot) and Yvonne and many other moments such as the "THANK – Thursday-drink," departmental trip together etc.

I would also like to mention my colleagues of ENP and to thank them for their support, social interaction and many dinners and drinks we had: Marjanneke, Harry, Jennifer, Judith, Hilde, Carolina, Elizabeth, Judith vL., Harry, Kim, Eira, Radhika, Sarah, Leah and Dorien Korbee. Special thanks to Dorien, you are one good-will, warm hearted person I ever came in contact with in my life. Dorien took me in her home during my stay in Wageningen during the 2010 winter when I was not availed appropriate accommodation. Thank you very much and extend my sincere thanks to Andreas as well as Normeel.

To the 2006/07 and 2010-13 Tanzanian and wider East African students whom in one way or another we interacted socially and academically: I will always be grateful to you as we shared many pleasant cook-in dinners and African dances which were instrumental in relieving all the academic and environmental pressures I was on at various times.

I acknowledge the office of Mwanza City Director, City Health department and the solid waste management contractors in Mwanza City whom they provided direct support without which the field research could not have been conducted. I would like to specially acknowledge Dr. F. Kimaro (Mwanza City Doctor-Health Department), Mr Kapizo and Kamenye for the provision of office accommodation and assistance during the field research in Mwanza.

To my family members; my brothers and sisters, uncles and aunties, cousins and nieces, my in-laws, relatives, friends and staff members of School of Environmental Science and Technology of Ardhi University, I thank you all for your continuous support.

To my ever loving mother (Bertha), thank you for your encouragement, prayers and belief in me. This achievement is the fruit of your sweat, tears, love and hope that you endured. It is my greatest hope that dad is looking from heaven and praising you for working so hard and getting your son to this level.

To my dearest boys Kevin, Andrew, Brian and Ethan. Thank you!!. I remember how Brian would ask "daddy are you coming this December or there is another December?" Andrew would be so happy to see an airplane fly over the house and would later ask me on the phone whether that is the plane that took me away. Kevin would say, what exactly are you studying that you are not finishing until I have gone to grade 2 and 3 and now grade 4?

I thank my lovely wife Devota, it must have been very tough to keep up with the boys on your own. Thank you for your love, continuous support, prayers, encouragement all these years with constant travelling to the Netherlands, Mwanza, regional and international conferences and workshops and long hours in the office. You have been my inspiration the entire time of study. Thank you my love.

Lastly, my gratitude is due to everybody who assisted me in any way during my studies. Not mentioning your name does not mean I don't appreciate your input but I am limited by space. "Nakushukuru sana" (I thank you very much).

Table of Contents

Abbreviations

ANAMMOX	– Anaerobic Ammonium Oxidation
BOD	– Biological Oxygen Demand
BOD_5	– Five-day Biological Oxygen Demand
CBOs	– Community Based organizations
COD	– Chemical Oxygen Demand
E-PRTR	– European Pollutants Release and Transfer Register)
FC	– Field capacity
GHG	– Greenhouse gas
GWF	– Global Warming Factor
GWP	– Global Warming Potential
HELP	– Hydrological Evaluation of Landfill Performance
IPCC	– Intergovernmental Panel on Climate change
IPCC	– Intergovernmental Panel on Climate Change
ISWM	– Integrated sustainable waste management
LandGEM	– Landfill Gas Emissions Model
LFB	– Landfill bioreactor
LFG	– Landfill Gas
LTP	– Leachate Treatment Plant
MCC	– Mwanza City Council
MMA	– Modernized Mixtures Approach
MR-TF	– Materials Recovery - Transfer Stations
MSW	– Municipal Solid Waste
NGOs	– Non-Governmental organizations
OFMSW	– Organic Fraction Municipal Solid Waste
OM	– Organic matter
PCC	– Post-closure care
PVC	– Poly Vinyl Chloride
SMLRM	– Stochastic Multiple Linear Reservoir Model
TANESCO	– Tanzania Electric Supply Company
TDS	– Total Dissolved Solids
TKN	– Total Kjeldahl Nitrogen
TZS	– Tanzanian Shillings
UNFCCC	– United Nations Framework Convention on Climate Change
UNC	– Unit of National Currency
UASB	– Up-flow Anaerobic Sludge Bed
UBAF	- Up-flow biological aerated filter
USEPA	– United States Environmental Protection Agency
UNFCCC	– United Nations Framework Convention on Climate Change
VFA	– Volatile Fatty Acids
VSS	– Volatile Suspended Solids

CHAPTER 1

Introduction

1.1 Background

Municipal solid waste (MSW) management in its broadest sense is concerned with the generation, on-site storage, collection, transfer, transportation, process recovery and disposal of wastes. It is a challenge in both highly industrialized western countries and developing countries. The increasing share of the population living in cities poses serious challenges to the provision of MSW management services by the cities/municipalities that are short of funds, deficient in institutional organization and interest, have poor equipment for waste collection, and lack urban planning (Rotich et al. 2006; Parrot et al. 2009). In most East African cities, including the cities in Tanzania, MSW management services is one of the most pressing environmental sanitation issues. A first aspect of flawed services is the lack of accessibility for a substantial part of the population. Waste collection systems are far from covering all communities, so that a large part of the waste remains uncollected causing large-scale negative health impacts and environmental nuisance. In most cities of Tanzania up to recently only about 40% of all waste that needs treatment and/or disposal is collected. The same situation is found in other East African cities of Uganda and Kenya (Kasozi and von Blottnitz 2010; Office of Auditor General 2010; KCC 2006). A second aspect is deficient treatment and disposal of the collected wastes. In many East African cities collected waste is simply deposited at dumpsites. This causes serious soil, groundwater, air pollution and health impairment and neglects possibilities for resource recovery, re-use and recycling (Kaseva and Mbuligwe 2000). In many cities in developing countries dumpsites need to be upgraded to engineered landfills in order to counteract the negative environmental and health effects. Engineered landfills are provided with protective measures against soil and groundwater pollution by leachate and have landfill gas capture in addition.

The present state of affairs in East Africa shows the failure of local government as the main provider of municipal services. One important failure factor is the use of large-scale systems which prove vulnerable to the political, economic and social instabilities that many developing countries face (Spaargaren et al. 2006). Furthermore, the tasks of local government as a service provider are complicated by rapid expansion of the urban population with its concomitant growth of the waste flows and handling costs (Powell et al. 2001), the poverty of parts of the un-serviced communities and the general lack of financial resources for environmental issues.

The importance of effective solid waste management has received wide recognition at international, national and community level. In the Agenda 21 of the UN Earth Summit in Rio in 1993 a number of specific objectives, programmes and targets has been spelled out to be attained. The particular agenda addresses environmentally sound management of solid wastes and sewage-related issues (Chapter 21). Nine years later during the World Summit on Sustainable Development in Johannesburg, the issue of environment was again high up in the ranking of priorities in the new millennium where Millennium Development Goal (MDG) # 7 calls for ensuring environmental sustainability.

New approaches to the challenge of MSW management now widely accept the need of integrated solid waste management systems with much more emphasis on waste reduction, reuse, recycling of materials as priority items (Al-Khatib et al. 2007) and maximization of resource recovery such as nutrients, chemicals and energy values of the waste. Waste reduction, reuse and recycling measures include creation of material recovery facilities and transfer stations which can divert waste components such as paper, plastics, metals and glass for remanufacturing into new products and the biodegradable organic portion can be recovered as feedstock for biological processes (Tchobanoglous et al. 1993, p. 543). Biodegradable organic wastes which constitute a large part of the MSW particularly in tropical developing countries can be treated by anaerobic digestion and composting or a combination of the two. Through anaerobic digestion a significant part of the organic material can be converted to biogas which can be utilized as source of energy. The produced compost can be beneficially used as soil conditioner (Tchobanoglous et al. 1993; Lens et al. 2004, p. 232, 234). In most East African cities demand for compost is limited due to presence of other soil conditioners readily available such as manure. Incineration is another possible treatment method. Through incineration of MSW energy can be recovered as heat and electricity. For East African cities incineration is deemed less appropriate than anaerobic digestion and composting due to the relatively high moisture content and low heat value of the wet wastes, the high initial costs and the high management and operational skills this technology requires. Therefore incineration will not be discussed further in this thesis. As the emphasis is now on recovery of resources, landfilling, even in the more environment-friendly form of sanitary landfills, has become the waste treatment method of last resort. In the European Union landfilling is largely restricted to materials for which no other destination than disposal can be found. Such materials are prescribed in the Council directive 99/31/EC on the landfill of waste. Existing old landfills are recovered, so that the useful materials and space can be reused. In the USA sanitary landfills are (still) widely used. One of the main reasons for the European restrictive policies apart from a focus on material recovery are high land prices.

In African cities as in many other developing countries some form of landfilling is practically still the only way of MSW treatment and disposal. In Africa land around cities becomes increasingly more expensive and siting of new landfills becomes more cumbersome, conventional landfills will become less and less popular. As a consequence, the road of integrated solid waste management has to be taken and new treatment and disposal techniques developed. Nevertheless, it may be expected that modern forms of landfilling will remain important in Africa on the short and middle term as the introduction of other more sustainable waste treatment and recovery methods probably will need considerable time. Therefore, in this thesis the innovative method of the landfill bioreactor (LFB) in combination with other treatment technologies is elaborated and assessed. The most significant feature of the landfill bioreactor is the recirculation of leachate, which may lead to increased degradation rates and lower organic matter concentrations in the leachate than in sanitary landfills. The developed methods intend to overcome the aforementioned disadvantages of the conventional sanitary landfills. The most significant expected strengths of the landfill bioreactor is the more rapid biodegradation of organic matter, a high yield of utilizable biogas and reduced land use.

1.2 Study context

In cities of the L. Victoria region the solid waste management situation is very similar to those described above for developing countries and Africa in general. Municipal solid wastes are only partly collected. Some is deposited on open dumpsites scattered through residential areas while much of it is hauled and dumped out of the city boundaries. This reflects the prevailing “out of sight out of mind philosophy.” As the larger portion of the MSW consists of biodegradable organic matter, this causes considerable environmental impacts through emissions of leachate and greenhouse gases (EAWAG 2006). The same holds for uncollected wastes accumulating in the streets, drains or crude open dump sites. The deficient collection is compounded by non-technological causes such as lack of appropriate planning, inadequate political will and governance, weak enforcement of existing legislation, as well as the absence of economic and fiscal incentives to promote good practice, and lack of analytical data concerning volumes and compositions of wastes (UNEP 2005). Concentrating on poor technology and lack of analytical data of the wastes determines the rationale for the general objectives of this research namely the development of technological interventions to alleviate the environmental problems and recover the nutrients and energy resources that can be derived from the wastes.

This study is part of a larger interdisciplinary programme – Partnership for Research On Viable Infrastructure Development in East Africa (PROVIDE) with a long term objective to help realize the Millennium Development Goals (particularly MDG7) by improving the sanitation and solid waste management in Tanzania, Kenya and Uganda via the operationalization and application of the Modernized Mixtures Approach (MMA). On a short term, the programme objectives are aimed at identifying the existing and potential arrangements, systems and modules for sanitation and solid waste management in East Africa and assess them with respect to their technological, institutional, social and economic dimensions while using and developing a set of key criteria and indicators for the PROVIDE programme; Develop an integrated data-base containing information on criteria measuring: (i) Ecological sustainability of the infrastructure, (ii) Accessibility for the poor, (iii) Flexibility, resilience and robustness of infrastructural systems (both technically and institutional/socio-political). (iv) Identify, design and assist in developing the most promising (set of) modules, arrangements and systems in sanitation and solid waste management in different local East African contexts, fulfilling the criteria of sustainability, accessibility and resilience. As part of the PROVIDE programme, this thesis is placed on the identification, design and assistance in developing optimal system options for treatment and disposal of MSW in the local East African context in a bid to improve solid waste management.

1.3 Research objectives and research questions

This thesis takes as point of departure the need of cost-effective, land-saving and energy producing waste treatment technologies for East African cities. Given the societal context of these cities landfilling will remain an important treatment and disposal option. Among the various sanitary landfill options the anaerobic landfill bioreactor (LFB) has been selected as a most promising technology, either as stand-alone system or in combination with certain pre-treatment technologies. Accordingly the main objective of this thesis is:

To have developed and described landfill bioreactor based municipal solid waste treatment systems suitable for East African cities.

This broad objective has been translated to the following main research question of this thesis:

Which are technically feasible and resource-recovery oriented landfill bioreactor configurations that could match the conditions of East-African cities?

In order to answer this main research question, the following sub-questions have been addressed:

1. What are the current conditions and practices of waste collection and disposal in East African cities? (chapter 2)
2. What are the processes and performance, in terms of waste degradation, landfill gas production and greenhouse gas emissions, of LFB systems working under the conditions of tropical regions? (chapter 3, 5)
3. How the leachate generated from LFB could be treated? (chapter 4)
4. How do LFB perform with typical Tanzanian MSW? (chapter 6)
5. What are the sustainable configurations of LFB applicable in the East African context? (chapter 7)

1.4 Elaboration of the research questions and applied research methods

In order to achieve the research question listed in section 1.3 a number of methods were employed. An assessment of current conditions of waste collection and disposal in East African cities was achieved on a field work through direct measurement method of sorting and characterization of the waste collected in and around the city at selected wards. The practice of waste collection and disposal was also diagnosed through collection of secondary data, physical observation, site visits and a workshop to verify the information collected.

A comprehensive desk study on numerous literature dealing with processes, steering parameters and performance in terms of biodegradation of waste in landfills and landfill bioreactors was also carried out. From this study, design considerations for suitable landfill gas collection systems and leachate collection and recirculation systems are established and used in the proceeding chapters. Furthermore, landfill gas models were also reviewed and a set of own innovative landfill gas models are developed and used. Leachate is an inevitable consequence of landfills, so a thorough study was also conducted on leachate production mechanisms, sanitary landfill and landfill bioreactor leachate characteristics as well as proposals for plausible treatment options.

Another method that was employed for achieving question 4 was an experimental work by simulating a pilot scale 2.5m high waste matrix in a square metre reactor to test how a LFB would perform in the prevailing environmental conditions of tropical countries filled with typical Tanzanian MSW. Operation and monitoring of this reactor entailed collection of leachate samples and performing in-situ and laboratory analyses of basic performance parameters.

Answers to question 5 are found as a result of the information collected during field work, literature review, developed innovative models for landfill gas production and the experimental work conducted. An outcome of this question is a proposal of four innovative LFG system options with varying configurations and appropriate leachate treatment options that are suitable and applicable in the East African context.

The work on these research questions, as reported in the chapters 2 until 8, can be seen summarized in the following conceptual framework (Figure 1-1).

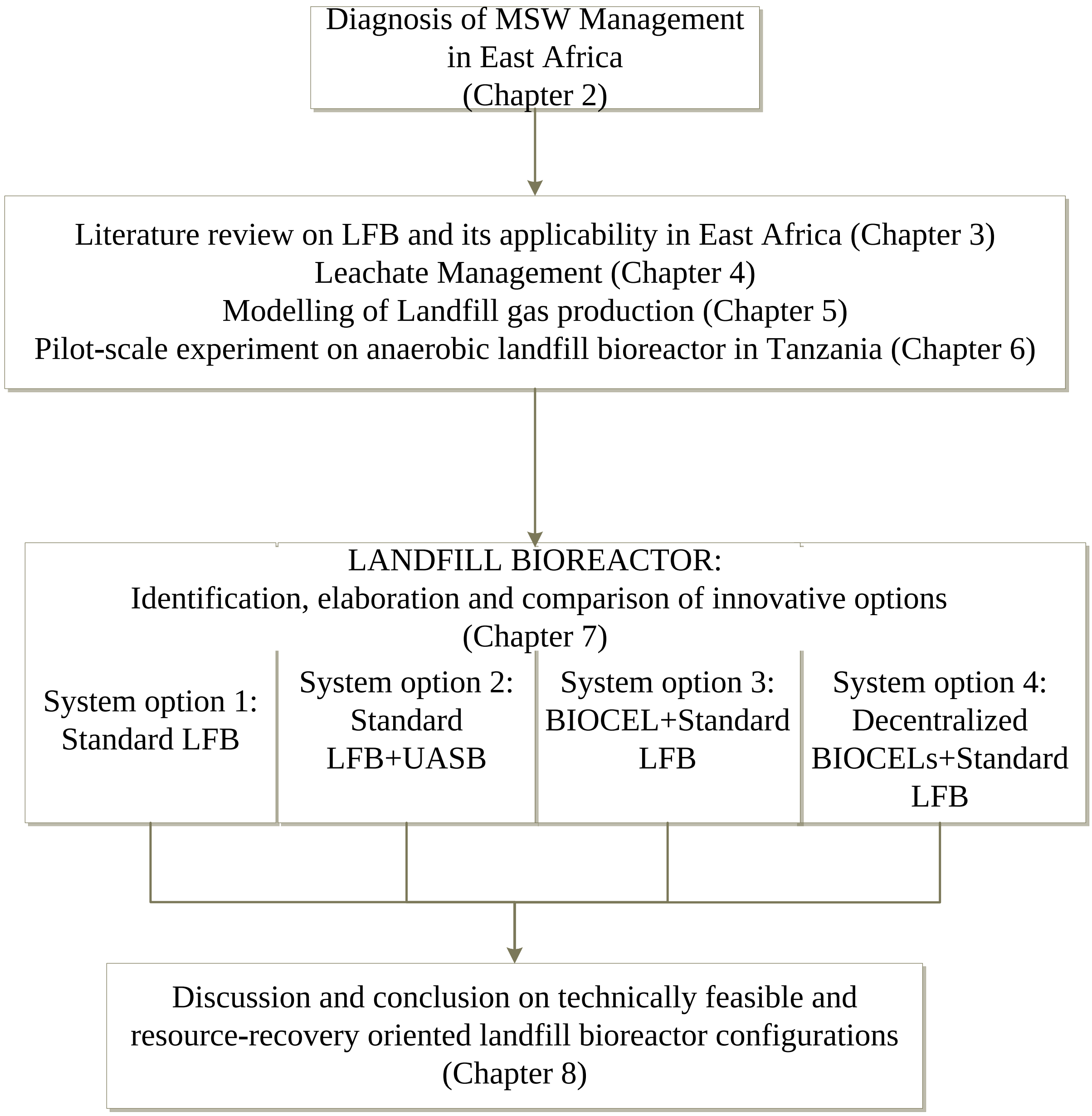


Figure 1-1: Conceptual framework

1.5 Outline of the thesis

This thesis comprises nine chapters. In the following, a short description is given on the chapter-content.

Chapter 1 is the general introductory chapter. It provides information on the problems of municipal solid waste management in Tanzania and East Africa and explains the rationale of the thesis. The chapter continues to describe the main research objective and the research questions addressed in the thesis.
Chapter 2 presents findings from an empirical research conducted in Mwanza City focusing on the current MSW management practice in Tanzania, waste characteristics and the waste generation rate.

Chapter 3 is a review of the LFB, general design of the LFB, the processes involved, steering parameters that influence operation of the reactor and closure and post closure issues to be addressed.
Chapter 4 looks at the available technological interventions of leachate for optimal management and treatment of the residues in the leachate after pre-treatment or recirculation in LFB, UASB reactor and BIOCEL process.
Chapter 5 presents an overview of models for calculation of the waste degradation and gas production in LFBs. The outcomes of this chapter are applied in chapter 7.
Chapter 6 assesses the performance of a pilot scale LFB in terms of the variations of the effluent leachate characteristics as an indicator of waste stabilization, the effects of leachate recirculation on COD removal, LFG generation rate and composition and leachate pre-treatment before re-circulation.
Chapter 7 reveals innovative LFB configurations aimed at optimization of energy recovery and suitable for the East African context
Chapter 8 presents the findings, reflects on the applicability of LFB in the East African context and gives conclusions and recommendations.
Chapter 9 is a summary of the main findings of the research

CHAPTER 2

Municipal solid waste management diagnosis of rapidly growing cities of East Africa[1]

2.1 Municipal solid waste management in developing countries

Effective municipal solid waste (MSW) management is an essential public service which benefits all urban residents. MSW management practices have undergone a dramatic evolution over the past thirty years in response to human health and environmental concerns (McBean et al. 2007). Since the early 1990s, many governments in developing countries have been showing a great concern in improving urban MSW management. This is because urbanization and rapid economic growth of cities have led to proportionate increases in waste generation with consequences to environmental health and cleanliness and to the demands on, waste collection, transport, treatment and disposal.

In developing countries and countries with economies in transition, waste management often emerges as a costly and complex activity that carries risks for both public health and the environment. To make matters worse, waste management usually has a low priority on the political agenda of such countries, as they are struggling with other stressing issues such as hunger, health problems, water shortages, unemployment and even civil war. In such situations, it is easy to understand why waste problems have a tendency to grow steadily (UNEP-IETC 2004). The challenge therefore for the municipal authorities in a bid to match the urbanization is to come up with innovative MSW management practices that are viable and sustainable i.e. satisfying short-term objectives without compromising on the long-term objectives (Kaseva and Mbuligwe 2005).

In the course of achieving proper MSW management, most formal Tanzanian efforts - like in many developing countries - focus on the collection and disposal activities, largely ignoring waste recycling and reuse possibilities. This approach is counterbalanced by a lively informal recycling and reuse sector with little or no recognition from the municipal authorities.

While most developing countries share this “collection to disposal” MSW management strategy, there are also innovative steps possible to improve this situation. Such practices have to be broad enough to be applicable at national level, and specific enough to address the characteristic needs of municipal solid waste systems on the local level. Establishing appropriate practices requires answers to crucial questions that can only be obtained by diagnosis (analysis and characterization) of the existing waste management system (Hristovski et al. 2007). Such an approach aims at the optimization of technologies and technological systems for waste handling. It uses an analysis along the three dimensions of flows, actors and the technological and socio-economic aspects (Spaargaren et al. 2006) as also attested in the integrated sustainable waste management (ISWM) concept (van de Klundert and Anschütz 2001).

[1] This chapter is partially published in proceedings of The 26th International Conference on Solid Waste Technology and Management as:
Municipal solid waste management diagnosis of rapidly growing cities: Case of Mwanza city, Tanzania. F. Salukele, S. Mgana, G. Szanto, J. van Buuren and W. H. Rulkens (2011)

Therefore, the main objective of this study is to find a basis for improved waste management in East Africa by diagnosing the current MSW management practice in Mwanza City in Tanzania one of the major cities of the country in the Lake Victoria region. More specifically, the activities carried out in this diagnosis include the making of an inventory for the current practices in waste collection and disposal in the study area (Mwanza) and the characterization of the collected MSW. Knowledge of waste characteristics and composition is indispensable for an appropriate choice of systems to manage the waste in a particular locality and it is in particular needed for the elaboration of suitable system options for Tanzania. It is important to be aware that definitions and classifications of solid waste vary greatly in the literature and in the profession (Tchobanoglous and Kreith 2002, p 1.2) The focus of this study is on MSW which includes all community wastes originating from residential, commercial and institutional sources, excluding wastes generated from agricultural activities, industrial processes, medical facilities and municipal services such as water and wastewater treatment systems.

2.2 Study area description

Mwanza City is the regional administrative headquarter of Mwanza region, one of 25 regions of Tanzania. It is the second largest city in the country after Dar es Salaam, located on the southern shores of Lake Victoria in Northwest Tanzania as shown in Figure 2-1. It is situated between 32° – 34° longitude East and 1° – 3° latitude South at an altitude of 1140 metres above the mean sea level. Mwanza experiences mean temperature ranges between 20 and 30^{o}C in the hot season and 15 and 18^{o}C in the cooler months.

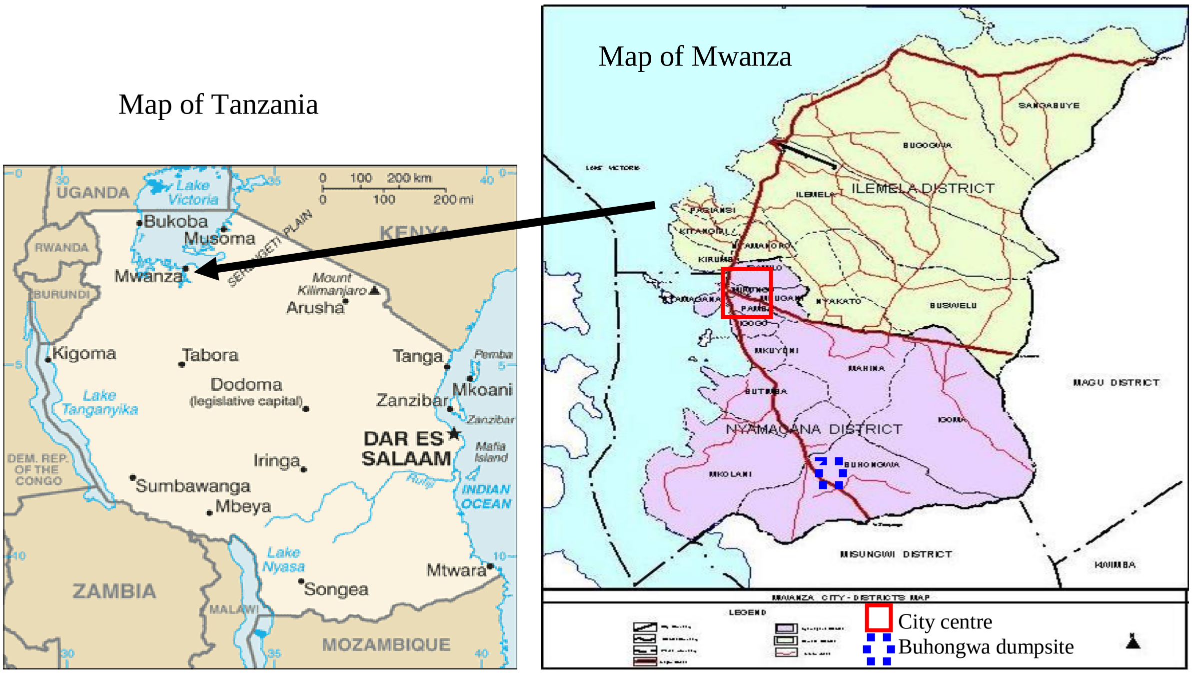


Figure 2-1: Map of Tanzania and Mwanza City showing the districts

The city receives between 700 and 1000 mm of rain per annum with two rain seasons: long rains from December to May and short rains from August to October. Mwanza City has two districts namely Ilemela and Nyamagana. The two districts are administratively divided into two divisions making 21 wards, 19 villages and a total of 523 streets. Out of the 21 wards, 14 are located in the urban area and 7 are rural. The city covers an area of 1325 sq.km of which

425 sq.km is dry land with undulating rocky hill areas and 900 sq.km is speckled with small islands in Lake Victoria. According to the 2002 National Census, Mwanza City has a population of 476,646 people with an average of 7 people per household and an annual growth rate of 3.2%. The annual rural to urban immigration amounts to almost 8% (Census 2003).

2.3 Methodology

In this study, four research methods were employed: desk study; interviews and questionnaire survey; field observation of practices and participation in MSW management activities and; waste characterization and quantification. The study was conducted in Mwanza City from October 2007 to March 2008.

The desk study involved literature review and other forms of secondary data gathering. Interviews and questionnaire survey were conducted mainly to obtain additional primary data and verify secondary documentation. The stakeholders interviewed were: Mwanza City Council – Health and cleansing department; Sustainable Mwanza programme – Office of Environment; one (1) Franchisee contractor; nine (9) Community Based organizations (CBOs) and five (5) Non-Governmental organizations (NGOs) and other relevant stakeholders – 11 (eleven) ward leaders, two (2) representatives from large solid waste generators (e.g. markets and small scale industries). Later on a workshop was conducted with all the major stakeholders to organize a feedback of the findings from the interviews, questionnaires and field observations.

Field observations were made at various waste generation and collection points and at the dumpsite site located in the outskirt ward of Buhongwa. Waste sorting and analysis involved direct measurements based on an output method (Sharma and McBean 2007). The output method for estimating the composition of the municipal solid waste stream was generally carried out at secondary collection points in 8 out of 14 urban wards. The 8 wards were selected based on the designated income levels which were low, medium and high income as obtained from secondary data of the city profile. The output method has numerous strengths which include provision of information unique to local planning for waste collection, recycling, treatment, and disposal and can be easily tailored to local needs (e.g. source type, generation characteristics, and seasonal variability). For the purpose of this study the following waste categories were distinguished:

- Food waste –food remains, fruit, fruit peelings, vegetables and other biodegradable waste;
- Grass/leaves –fruit and banana packaging materials, tree trimmings, broken furniture/timber, yard and garden waste;
- Waste paper –printer paper, newsprint, card board, wrapping paper, boxes, etc.;
- Plastics –PE bags, PETE, LDPE and HDPE bottles, containers and PVC based materials;
- Textiles – pieces of clothes, rugs, pieces from tailoring marts (rags);
- Metals – ferrous, non-ferrous metals, bi-metal and aluminium cans;
- Glass – glass bottles (clear, green, brown) and jars for food and beverages;
- Ash – ash mixed with burnt out charcoal;
- Sand, stones and fine earth – from street swept refuse;
- Other wastes - batteries, electrical and/or electronic waste, dead large-bodied animals, etc.

A detailed sampling protocol was outlined whereby (wet) weight fractions of the waste components were quantified. Waste sorting and analysis were made during one week for each of the two distinct seasons (i.e. in November during the dry and in February during the rainy season). As the dumpsite of Buhongwa has no weighbridge, the sampling and sorting had to be done at secondary storage sites (i.e. at the transfer stations and at the waste collecting skip buckets). At these sampling points, waste brought in by primary collectors in amounts easily carried by one or two persons was sorted and placed in plastic bags and then weighed by aid of a spring balance. The primary collection of waste was done continuously throughout the day whereby a data collection sheet was used to record the waste components as they came in. These data were later aggregated to the total waste brought to the sampling point.

Accordingly, the daily amount of waste collected per ward and reaching the secondary storage facilities was established. The specific weight of the waste was aggregated as the sum of unit weights of each waste category as obtained during the one (1) week long survey in the two (2) distinct seasons of the year that the city experiences. There were no outstanding differences in waste collection between the two seasons therefore an average value was established to represent the characteristics and composition of wastes generated in the city for the entire year. In this study per capita generation rates were calculated by taking the total amount of waste collected divided by a project population of Mwanza city of the year 2007. The number of people in 2007 was established by taking the 2002 census and projecting it to 2007 using an annual population growth rate of 3.2% as reported by the National Bureau of Statistics (NBS 2006).

2.4 Results and Discussion

2.4.1 Overview of the Mwanza City solid waste management system

There exists an old legislation controlling the local government's powers and responsibilities, namely the Local Government (Urban Authorities) Act of 1982. This legislation provides the local authorities with the responsibility "*to remove waste and filth from any public or private places and provide and maintain public dustbins and other receptacles for the deposit and collection for solid waste.*" Furthermore, according to the same Local Government Act 1982 and also according to the Environmental Management Act section 38-1(a) the City councils shall be responsible for the proper management of the environment with respect to the area in which they are established. Section 114-2(b) of the Environmental Management Act, local government authorities shall, with respect to their areas of jurisdiction, manage solid waste generated in accordance with sustainable plans produced by respective local government authority (EMA 2004). Following this Act, Mwanza City Council (MCC) has been the responsible authority for providing MSW management services to its residents over the years. Since it is a public service by a government institution, it was being provided without charges to the citizens. At the city council level MSW management was and still is administered through the City Health Office (CHO) by a subsection known as Health and Cleansing in collaboration with local government officials (i.e. ward and street leaders).

The roles and responsibilities of the Health and Cleansing subsection are: street sweeping, mowing and cleaning of public spaces, unblocking and cleaning of storm drainage channels and most importantly collection and transportation of the collected waste to designated transfer stations and disposal site. Currently the subsection is also tasked with the collection of waste from onsite sanitation facilities on a commercial basis (i.e. hiring of cesspit emptier trucks). Privately owned cesspit-emptier trucks also carry out the same business within the city. Both responsibilities of solid and liquid waste management weigh heavily on diminutive municipal resources. As a consequence, the MSW problems in Mwanza city are on the rise. The main sources of solid wastes of Mwanza city are:

- Residential sources – households (low, medium and high income areas all mixed);
- Commercial sources – hotels, restaurants, market places, shops, bus, railway station, etc.;
- Institutional sources – schools, colleges, offices, banks, etc.;
- Industrial premises – light and heavy industries.

Table 2-1 below presents an inventory of such activities as of 2006. Apart from the listed licensed businesses, there are many informal commercial and industrial activities as well. The diversity of the businesses in the city poses a serious challenge to the authorities to manage the wastes generated. The waste from these sources are not collected separately but mixed.

Table 2-1: Inventory of business activities in Mwanza City

BUSINESS TYPE	COUNT	BUSINESS TYPE	COUNT
Bars	300	Abattoirs	1
Hotels	34	Markets	9
Restaurants	76	Fish Mongers	48
Local brew shops	50	Pharmacies	28
Guest Houses	169	Medical stores	450
Groceries	135	Barbers shop	60
Retail &Wholesale Shops	2528	Stationeries	142
Bakeries	10	Milling Machines	110
Butchers	150	Wood works	61

Source: (MCC 2006a)

The waste management system of Mwanza city is depicted in Figure 2-2. Waste in the city is collected from the sources by contractors formally recognized by the MCC and hauled to secondary storage points where there are communal chambers, skip buckets or transfer stations. From the secondary storage points, waste is transported by MCC and franchisee trucks to the MCC owned dumpsite in Buhongwa. This dumpsite is located 18 km from the city centre and extends over an area of 232.5 acres (≈94 hectares).

It should be noted that the only franchisee operating in Mwanza city also has independent contracts with other institutions such as Bugando referral hospital whereby the waste collected from the hospital is also hauled to the Buhongwa dumpsite as illustrated by the dotted region in Figure 2-2.

Wastes from industries are also transported to the dumpsite by trucks contracted by the respective industry. Scavenging activities for recoverable materials such as plastics, metals, textiles, paper boxes etc. were observed at the secondary storage points and at the dumpsite. The quantity of materials could not be established because the activity is informal and none of the scavengers was ready to volunteer information regarding where the destination, the selling price and the future use of the collected materials.

The responsibilities of waste management weigh heavily on the municipal resources and the MSW problems in Mwanza city were eminent. The level of performance of MSW provision by MCC in 2002 was rated at approximately 40% collection efficiency (MCC 2006a) which was very low, so that huge amounts of waste were left uncollected and unattended in the environment.

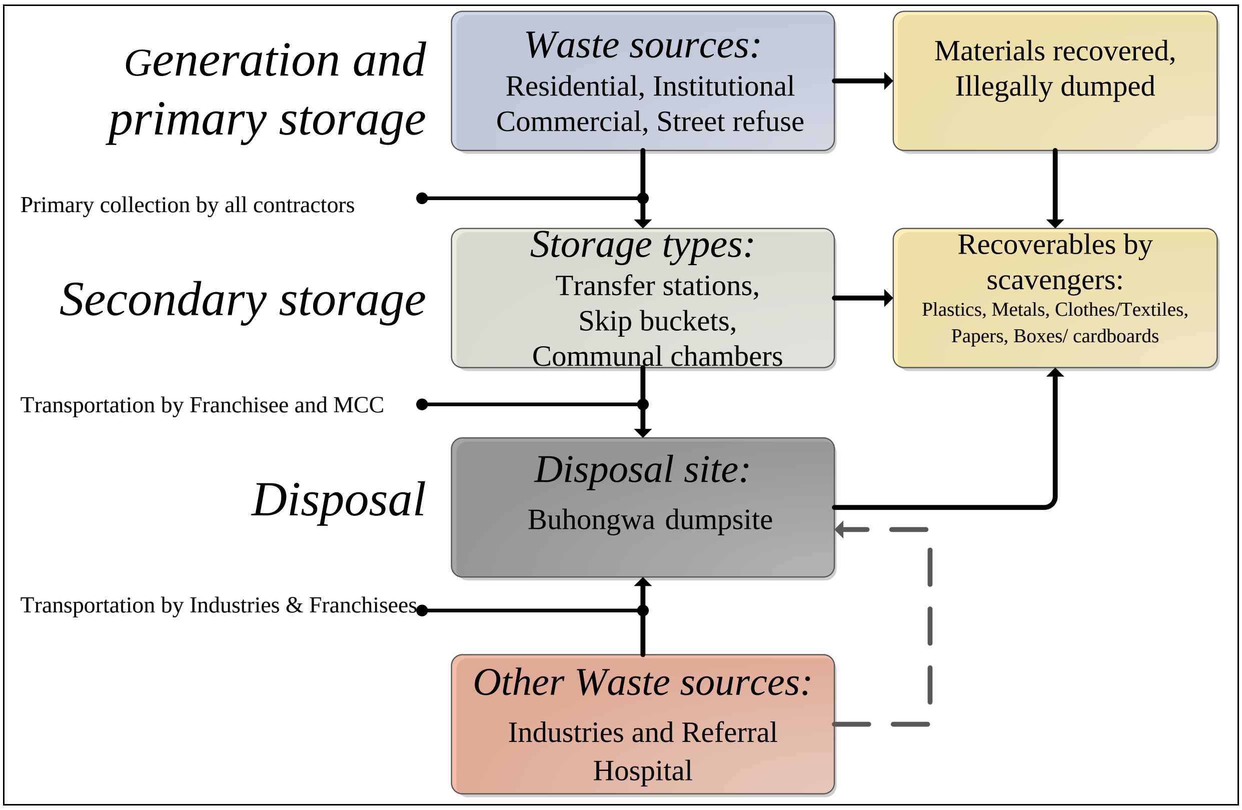


Figure 2-2: The existing MSW management system of Mwanza City

The uncollected refuse accumulated in drains, on open land and served as breeding areas for disease vectors. In order to remedy the situation an approach of contracting out collection and transportation services to private solid waste contractors was employed.

2.4.2 Privatization of MSW management services

For the purpose of achieving sustainability of MSW management and accessibility of the services to all citizens, MCC adopted the approach of privatization in the year 2002. The involvement of the private sector in MSW management according to the tender documents had the following objectives:

a) To involve MSW generators in the efforts of keeping their respective areas clean;
b) To reduce the responsibility of the city council on the provision of service in order to concentrate more on monitoring and supervision of the cleanliness activities;
c) To achieve a long term and sustainable solution to the MSW management problems;
d) To create employment opportunities;
e) To increase collection efficiency of waste thus reducing probable spread of diseases.

The objectives were comprehensive with the overall mission of protecting the environment and enhancing the public health of the city dwellers. They were also geared towards community participation in environmental protection.

In the privatized MSW management regime, the main stakeholders are the local government, the service beneficiaries, the contractors and the central government. The local government which for this case is the MCC is the employer of the contractors who operate under its jurisdiction.

At MCC level as mentioned earlier, coordination is through the City Health Office (CHO). The CHO monitors the performance of contractors and sets the MSW management performance standards and the fees that the contractors can charge for their services. The beneficiaries of the contractors' services are obliged to pay service fees. In case of dissatisfaction regarding service quality, they can lodge their complaints to their local government through Ward health officers. They are responsible for overseeing the provision of services at their respective wards and report to the CHO.

MSW management contractors are those CBOs, NGOs and franchisees who win the tenders based on their attributes and capacity to provide the service. These attributes include possession of an able and experienced workforce, equipment, good financial standing and commitment to the service. The latter is judged indirectly by considering whether MSW management is the primary activity of the contractor or it is one of many other commercial activities.

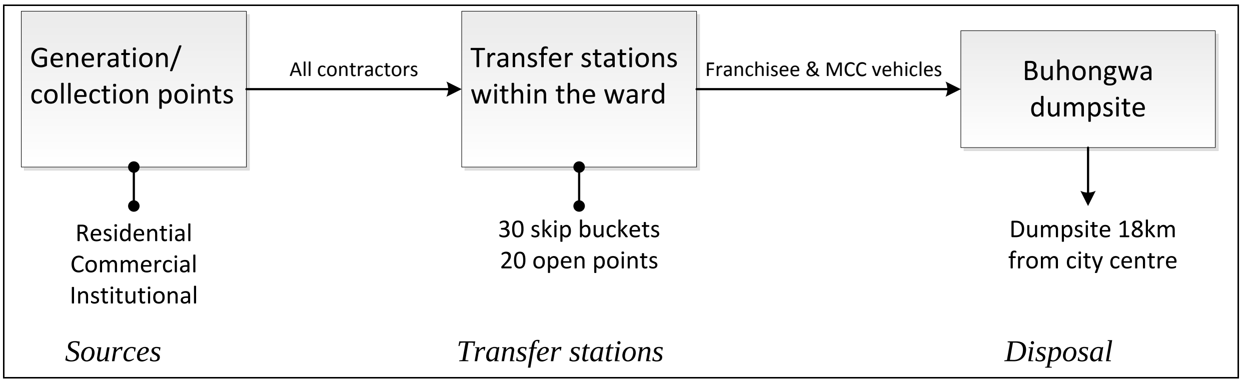


Figure 2-3: Roles and responsibilities of MCC and the contractors in Mwanza City
(Survey conducted in November 2007 and March 2008)

The central government's role is the financing of the MCC to purchase waste transporting vehicles as wheel loaders, excavators, refuse collection trucks and also fuel. The central government also plays a role as the overall overseer of the MSW management service efficacy. The existing roles and responsibilities and attributes of the stakeholders in Mwanza City are as depicted in Figure 2-3.

In the privatization process, the city council floated tenders after which thirteen (13) contractors were selected to provide the MSW management services to eight (8) wards out of the twenty one (21) wards of Mwanza City. In 2006, two (2) wards were added and in 2007 four (4) more wards were also included and new tenders were floated, thus making fourteen (14) wards of urban and peri-urban served by fifteen (15) private contractors. It should be noted that there are two wards that are split into two in terms of MSW collection only but not administratively (i.e Nyakato A and B and Nyamanoro A and B). Out of the 15 contractors, one (1) is a franchisee, nine (9) are CBOs and five (5) are NGOs. Table 2-2 shows a list of wards and their respective contractors. Since the inhabitants in rural wards mostly use resources of biological origin harvested from the surrounding environment, the waste is disposed by burying in pits or used as animal feed.

Table 2-2: Wards in Mwanza City and their assigned waste collection contractors

Ward	Contractor name	Contractor type
Pamba	Prima bins	Franchisee
Nyamagana	Prima bins	Franchisee
Isamilo	ETIA	CBO
Igogo	Kinyagesi B	CBO
Igoma	MKUA	CBO
Kirumba	QED	CBO
Mbugani	Maendeleo Mbugani	CBO
Mkuyuni	Himaja	CBO
Nyakato A	UZOTA	CBO
Nyamanoro A	Muungano wa Wajane	CBO
Pasiansi	PATUMA	CBO
Butimba	Boresha Mazingira	NGO
Kitangiri	Charity Organization	NGO
Mirongo	CHASSAMA	NGO
Nyakato B	TUFUMA	NGO
Nyamanoro B	Maendeleo Mkudi	NGO

Source: Survey conducted in November 2007

2.4.3 Contracts

Contracting is a viable means of securing service as long as it is possible to adequately describe outputs anticipated from the contract (Cointreau Levine 1994). The MCC prepared tender documents which were circulated widely among capable firms to compete for contracts. Successful firms were given contracts which agreed on the following outputs: a) MSW collection in the wards, b) transportation of the collected waste to transfer stations and to the dumpsite, c) sweeping of roads and pavements, d) mowing and cleaning of public spaces, e) removal of sand and unblocking of storm drains and f) collection of fees from households and commercial premises. There are two types of contractual agreements between MCC and the private firms. These are discussed in the following sections.

2.4.3.1 The CBO and NGO contracts

The CBO and NGO contractors receive a fee paid by the city council for the services they provide as per the contractual agreement. During the study it was observed that the contractors receive revenue from two sides: (1) a contract fee from the MCC for cleaning storm drains and sweeping the streets and roads at a rate of TZS 1000 (approximately 1 US$) for every 300 metres; (2) from the waste generators for waste collection. According to the contract they are supposed to pay the MCC 5% of all revenue collected. Furthermore, it is in the contract that the contractors are required to transport the collected waste to the dumpsite but the city council assists in the transportation of the waste to the dumpsite at a fee of TZS 8000 (equivalent to approximately 8 US$) per seven ton of waste. Consequently, depending on the amount of waste collected and the collection frequency, the city council deducts the respective transportation charges from the contract fee.

2.4.3.2 The franchisee contractors

The franchisee receive its revenue from the waste generators. Contrary to NGOs and CBOs, the franchisee contractors receive no payment from the city council. It is their sole duty to collect all wastes, sweep the roads and storm drains and keep public spaces clean in their contracted areas, transport the waste to the dumpsite and collect a refuse collection fee. The major difference between this type of contract and that of CBOs/NGOs is that the franchisee must transport the waste to the dumpsite and is responsible for its own earnings. The involvement of the MCC is only supervision and monitoring of the rendered services.

2.4.4 Performance assessment of privatization of MSW Management services

According to the Preventive Health Services profile (MCC 2006b) the amount of waste collected for disposal before privatization was less than 40% of the wastes total waste collected. After privatization in 2002, the collection efficiency rose to 61%, and gradually continued to rise to 78%, 84% to 88% in 2004, 2005 and 2006 respectively. According to the CHO, the collection efficiency was still 88% in 2007 when this study was conducted,. However, during this study, a different value of collection efficiency was evaluated as discussed in section 2.4.6.

Privatization also created a livelihood for the urban poor by providing employment and a business activity for the community. As mentioned earlier, Mwanza city is served by 1 franchisee, 9 CBOs and 5 NGOs in the urban wards. The interviews with authorized spokesmen of these contractors revealed that privatization of MSW management services led to employment of 273 people among whom 80 males ranging between 25 and 35 years of age and 193 female ranging between 20 and 55 years of age. In turn the CBOs and NGOs were said to have been able to make a small margin of profit though none of the interviewees was able or willing to tell how much this profit was.

2.4.5 Waste collected and generation rate in the city

Waste streams were sorted and analyzed in eight (8) selected wards as depicted in Table 2-3. Butimba, Igogo and Mirongo count as low income wards, Kirumba, Nyamanoro and Pasiansi are middle income, while Nyamagana and Pamba are high income wards located in the central business district. Table 2-3 illustrates the income level of the wards, their respective population and the amount of waste collected daily.

From the direct measurement method the average amount of the waste collected in the surveyed wards was found to be 102.8 tons/day with a per capita generation rate of 0.32 kg/day (i.e. an average of 12.8 tons/day per ward). Therefore, the amount of waste to be collected from all 14 urban wards of Mwanza City whose population is about 562,500 people is about 180 tons/day.

The prediction of waste generation plays an important role in planning of MSW management. Traditional forecasting methods (static models) for solid waste generation frequently count on the demographic and income related waste generation rate on a per-capita basis. In this study the status of the wards in terms of their income levels was used as secondary data that was gathered in relation to waste generation.

Table 2-3: Ward income level, population, amount of waste collected daily and the per capita generation rate in the selected wards

Ward	Status of service area	Population (2002)	*Population (2007)	Waste collected (kg/day)
Butimba	Low income	26983	33786	4571.5
Igogo	Low income	38812	48597	6761.2
Mirongo	Low income	17406	21796	35867.2
Kirumba	Medium income	34867	47102	13752.1
Nyamanoro	Medium income	47824	64605	7618.9
Pasiansi	Medium income	32051	47378	14091.8
Nyamagana	High income, CBD	10646	13330	5779.4
Pamba	High income, CBD	31489	39429	14292.1
		TOTAL	**316023**	**102734.2**

Population 2002 is according to National census conducted in 2002
*Population 2007 is projected using a growth rate of 3.2% as established by the National census

Implementation of the traditional forecasting method requires collecting thorough socio-economic and environmental information before the forecasting analysis can be performed (Kassim and Ali 2006). A way of determining the total collection of wastes, i.e. the sum of wastes fated for disposal, is to multiply the per capita rate of generation by the population in the generation area. Per capita generation rates are generally more difficult to predict than population projections. The average per capita generation rates were established based on the total amount of waste collected divided by the population of the collection areas.

The average per capita generation rate found during this study, 0.32 $\pm$ 0.06 kg/cap/day is comparable with previous studies such as JICA (1996), Kaseva and Mbuligwe (2005) and the study conducted by the Inter-American Development Bank (2003) in (Zuilen 2006).

It is generally assumed that in communities with a higher income tend to generate more waste but that is not the same for Mwanza city. The amount of waste generated and eventually collected in each ward was varying without the influence of the income level of the ward as illustrated in Table 2-4. The middle income wards exhibited the lowest per capita generation rate and not the high income wards. This is a common phenomenon in cities of developing countries that not all socio-economic factors have an impact in the waste generation rates. This is mainly because even in the low income wards there are waste generating activities. For instance, at Mirongo which is a low income ward there is a fish market and a landing bay for passenger and cargo ships whilst in Kirumba, a middle-income ward, there is the City's largest fish market but most fish are taken to the processing industries at Igogo which is in the low income ward category.

Table 2-4: Ward income level, population and the per capita generation rate in the selected wards

Status of ward	Population (2007)	Per capita generation (kg/cap/d)
Low income	104179	0.45
Medium income	159085	0.22
High income, CBD	52759	0.38

The larger portion of MSW in Mwanza city consists of biodegradable organic matter. From Table 2-5, the organic waste collected per day in the eight surveyed wards adds up to about

86 tons/day which is equivalent to 84% (i.e. 46.2 % food wastes, 37.6% grass/ leaves) of all collected wastes. This waste being mixed and putrescible, needs attention in terms of treatment and disposal. Recoverable materials amenable for recycling and reuse such as papers and cardboard boxes, plastics, metals and glass account for about 14 tons (14%) of wastes collected. Other materials such as e-waste, batteries, ceramics, etc. amount to around 2.7% equivalent to 2.7 tons/day.

Table 2-5: Type and amount of waste collected in the surveyed wards

Composition	Food	Grass/leaves	Paper	Plastics	Textiles	Metals	Glass	Ash	Sand	Other
Amount (kg)	47422	38763	7517	2861	1491	961	977	1139	1524	81
Distribution (%)	46.2	37.6	7.3	2.8	1.5	0.9	1.0	1.1	1.5	0.1

Source: Survey conducted in November 2007 and February 2008

Figure 2-4 show the percentage distribution of waste composition collected in the city and Figure 2-5a and 2-5b shows the typical waste that reaches the dumpsite.

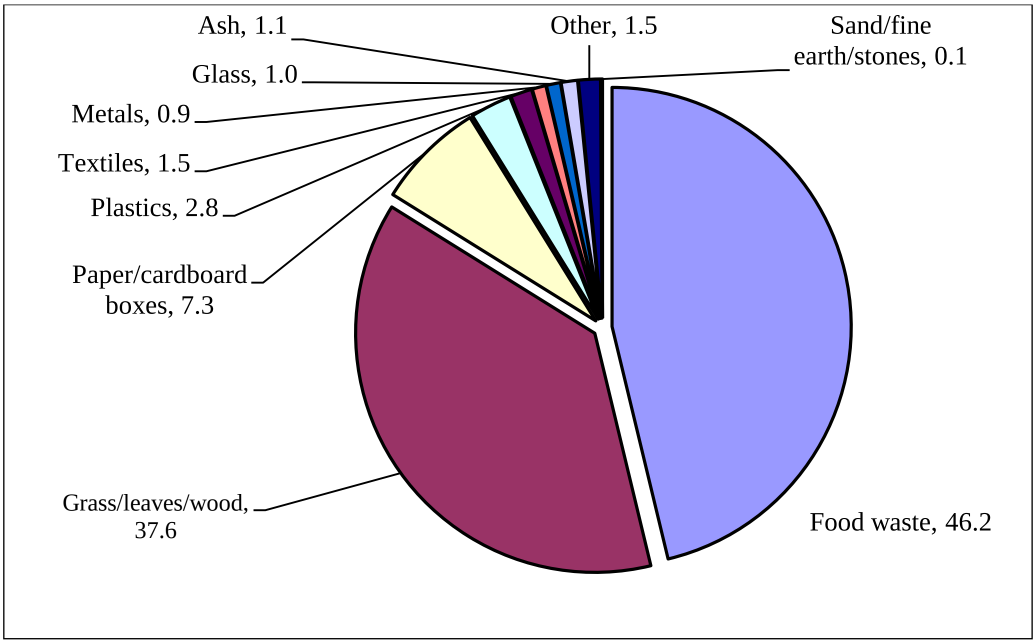


Figure 2-4: Percentage distribution of waste composition collected in the city
Source: Survey conducted in November 2007 and March 2008

Figure 2-5a: Typical composition of waste at the Buhongwa dumpsite
Source: Survey conducted in November 2007 and March 2008

Figure 2-5b: Typical composition of waste at the Buhongwa dumpsite
Source: Survey conducted in November 2007 and March 2008

2.4.6 Waste collection and transportation

MCC owns two skip loaders of 7 tons carrying capacity (during the study one was defective), four 7 tons side loaders (only two in good working condition) and 30 skip buckets of which about 70% are old and dilapidated. For carrying out landfilling activities at the dumpsite site MCC also owns 1 excavator in good working condition and 1 heavy duty wheel loader which during the study was defective. For collection of waste from onsite sanitation facilities, MCC has two cesspit emptier trucks. The franchisee owns three tipping trucks of 5 tons carrying capacity each. With all collection vehicles working at full capacity, the MCC and franchisee trucks makes a total of 27 trips per day carrying waste to the dumpsite. The dumpsite has no weigh bridge so the amount of waste reaching the dump is calculated based on an equivalent tonnage of the same truck type weighed in Kinondoni district in Dar es Salaam, Tanzania (i.e. the actual capacity see Table 2-6).

Table 2-6: MSW collection capacity and daily amount transported

Truck	Trucks (number)	Carrying capacity (tons)	Actual capacity* (tons)	Trips (trips/day)	Amount (ton/day)
Skip loader	2	7	6	3	36
Side loader	4	7	4.5	3	54
Five-ton tipper	3	5	3	3	27
				Total	**117**

* Actual capacity – The actual amount of un-compacted waste carried by such trucks. Figures adopted from Kinondoni district strategic plan for improving solid waste delivery (Kinondoni 2006).

From Table 2-6, it can be inferred that the daily amount of waste collected by the MCC trucks and reaching the Buhongwa dumpsite is 117 tons which is equivalent to 65% of the total waste collected in Mwanza city. From field observations and interviews carried out during this study, an estimated 20% of the waste collected is recovered by scavengers at the secondary storage points but it was not possible to find out the amount recovered (i.e. qq tons/year) by the scavengers at the dumpsite. 15% of the waste is left at the transfer stations. Figure 2-6 is a summary of the current practice of waste collection and disposal of Mwanza city.

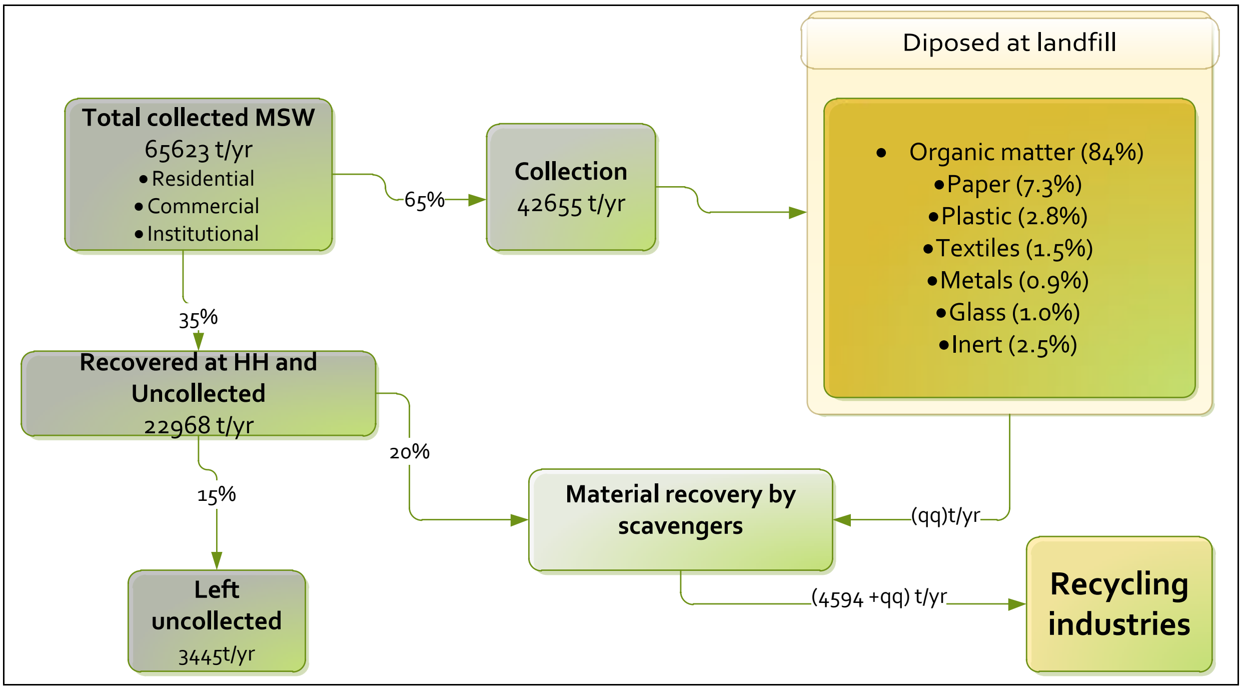


Figure 2-6: Estimated flows of collected MSW in Mwanza city

2.4.7 Waste disposal

The only method of disposing of waste in Mwanza City is landfilling, in fact uncontrolled dumping. The dumpsite is about 18 km from the city centre and occupies 94 ha sitting on a natural depression. At the moment of this study the Buhongwa dumpsite was used for about two years already. Before the commissioning of the Buhongwa dumpsite, waste in the city was dumped in an area called Nyakato located 10 km from the city centre. Due to numerous environmental concerns raised by people residing near the dumpsite and other reasons which include encroachment and development of human settlements and industries in Nyakato, that dumpsite was forced to close.

The Buhongwa dumpsite is not completely fenced and has no weigh bridge. At the entrance there is a gate whereby waste transportation trucks are registered and the weight of waste is estimated by the type of truck. Filling of waste is in cells not clearly separated and not well planned thus the average height of the dumped waste is very low not more than 1m. This is low as the area over which the waste is spread is large. If the density of the waste is assumed to be 0.5 ton/m^3 and an assumed maximum average dumpsite height of 10 m, at the present rate of 180 tons/day disposal (i.e. all waste collected from 14 urban wards) the Buhongwa dumpsite could last for ≈72 years.

However, the city council owns an excavator but running costs of the vessel has been a huge burden on the meager financial resources of the city thus not effectively used. The city council also own a heavy duty front wheel loader but for extended periods of time including the time during this study, it has been defective. The dumpsite has no bottom lining so leachate can percolate to the subsoil in an uncontrolled way with a potential to ground water pollution. The dumpsite has no gas collection or recovery system.

The waste load is tipped almost anywhere within the site and there are a number of waste pickers and animals and birds scavenging at the site. In some parts of the dumpsite waste was observed to be burning. The burning can result into development of hazardous gases (which can be explosive) and foul odour. Figure 2-7 is a combination of pictures showing the situation at Buhongwa dumpsite during this study.

Figure 2-7: Combo picture of the entrance, dumped waste, birds and smoke and leachate

2.5 Conclusion

Diagnosis of municipal solid waste management is important because it contributes to identification of problems, characterization of waste systems, quantification of waste generation and the data obtained could: (1) assist characterization of the waste management system; (2) assist in selecting and designing of appropriate technical solutions in order to improve the current and future situations.

MSW management usually has a low priority on the political agenda of developing countries, as the governments are dealing with other pressing issues such as unemployment, illiteracy levels, health problems, water shortages, lack or inadequate communication and energy infrastructures, industrialization and to keep up with the globalization. In such situations, waste problems have a tendency to grow steadily. And with diminutive resources allocated to municipal authorities, services levels of environmental concerns such as waste management tend to deteriorate.

In order to curb the municipal solid waste management problems particularly related to collection, City authorities have opted for privatization of the services which means that commercial and civic organizations are given a role. This is the case for Mwanza city where solid waste management services have been contracted out. After privatization, the waste management system of Mwanza city was characterized by constructive cooperation of the public (city council) and the private sector (contractors). The private sector includes NGOs, CBOs and one commercial franchisee that is specialized in primary collection activities (i.e. from households, collection points and other sources to skip buckets and/or transfer stations). The MCC along with the franchisee transports wastes from the skip buckets, transfer stations and illegal mini dumps (i.e. open spaces, roadside etc.) to the final dumpsite. This approach of privatization has been reported in Mombasa, Kenya (Henry 2006), Dar es Salaam, Tanzania (Kassim and Ali 2006). Calcutta, India and Kumasi, Ghana in (Kaseva and Mbuligwe 2005) and from a more recent study in Kampala where NGOs and CBOs are now full players in provision of services related to sanitation and solid waste management (Tukahirwa 2011).

The MCC together with the solid waste contractors are required to collect a projected 180 tons of waste in 14 wards of Mwanza city. However, only approximately 117 tons/day (65%) is the amount of waste transported to Buhongwa dumpsite and an estimated 20% is recovered by scavengers. The 65% collection is progress made compared to the 40% collection efficiency that the city council was capable of before privatization. It is the involvement of private sector that has helped to achieve the aforementioned collection. Formalization of waste pickers could reduce the amount of waste recycled so that less waste has to be landfilled. Major factors affecting collection of waste in Mwanza city are inadequate and dilapidated facilities and equipment, especially vehicles. The long distance to the dumpsite located at 18 km from the city centre makes it difficult to transport all the wastes with the limited number of vehicles.

From the direct measurement of waste, the daily average amount of waste collected in the surveyed wards was found to be 102.8 tons and for all 14 wards is 180 tons/day. The average MSW generation rate (after collection) found during this study, 0.32 $\pm$ 0.06 kg/cap/day is comparable with previous studies conducted in Dar es Salaam – Tanzania such as JICA (1996), Kaseva and Mbuligwe (2005) and the study conducted by the Inter-American Development Bank (2003) in (Zuilen 2006) whereby the domestic waste generation rate established ranged from 0.3 to 0.7 kg/cap/day.

According to the study component on characterization of the collected waste 84% of all wastes is organic in nature while 14% is amenable to recycling and reuse such as papers and boxes, plastics, metals and glass whereas other materials such as e-waste, batteries, ceramics, etc. are 2.7%. Without proper attention to the biodegradable fraction of waste such as appropriate landfilling technologies of the waste, environmental pollution, health and degradation implications will be imminent.

The characteristics of waste of Mwanza city as presented in this chapter are more or less representative of the waste that is generally generated in Tanzania and East Africa with small differences. Waste generated in Dar es Salaam, Tanzania is 3000 tons/day of MSW composed of about 65% organic waste, 17% recyclables (textile, metal, paper, glass) and 18% inert (organic and inorganic) according to (DCC 2004). Nairobi city, Kenya generates 3100 tons/day whereby 51% organic, 38% recyclables and 11% inert with a generation rate of 0.65kg/capita/day (Kasozi and von Blottnitz 2010). According to Office of Auditor general (2010) in Kampala city, Uganda out of 1,200–1,500 tons of MSW generated per day and the waste composition is 74% organics, 25% recyclables and 1% inert materials (KCC 2006). Like Mwanza, Nairobi has one designated waste disposal site, an open dump located in

Dandora area about 7.5 km south east of the city centre. Kampala city has sanitary landfill at Kitezi and Dar es Salaam city at Pugu kinyamwezi has a designed sanitary landfill but a controlled dumpsite is implemented instead. From all the cases cited in this subsection, there are opportunities for: enhanced stabilization of the organic fraction of the waste which is clearly the largest portion of the generated waste; potential for landfill gas recovery; reduced leachate treatment potential. This calls for an urgent need for improving the disposal practices that these East African countries are carrying out.

CHAPTER 3

Literature review on landfill bioreactors and applicability in East Africa

3.1 Introduction

This chapter presents a literature review of the fundamental processes, design and operation of landfill bioreactors. It aims to generate the insights to make well founded choices about the introduction of the landfill bioreactor technology in East Africa.

Historically, solid wastes such as MSW have been buried in the soil, which is a primitive form of landfilling. Major concerns of landfilling of wastes are the leachate escaping from landfills that can contaminate soils, aquifers, and surface waters and escaping landfill gases that contribute to the global warming effect (El-Fadel et al. 2002; Valencia 2008). Therefore, the air and water emissions from landfills must be monitored, controlled, and treated for a long time (Pohland 1980; Barlaz et al. 1990; Townsend et al. 1996; McCreanor and Reinhart 1999; Pacey et al. 1999; Yuen et al. 2001; Mehta et al. 2002; Reinhart et al. 2002). One idea that has gained significant attention in the past three decades in a bid to address the major concerns of conventional landfills is operating them as a bioreactor thus Landfill bioreactor (LFB).

3.1.1 Landfill bioreactor terminology

A LFB has been defined in various researches and reports as a sanitary landfill in which enhanced microbiological processes stabilize the readily and moderately decomposable organic waste constituents within a period of 5 to 10 years (Pacey et al. 1999; Reinhart et al. 2002). It is a MSW landfill, or a portion of a MSW landfill, where leachate, sometimes combined with additional liquids, is added in a controlled fashion into the waste mass (often in combination with recirculating leachate) to bring the moisture content of the degrading waste to at least 40% to accelerate the anaerobic biodegradation of the waste (Townsend et al. 2008). The operating procedures of a LFB are adjusted from those used at conventional landfills to quickly initiate the decomposition of the waste. Through its high conversion rates the LFB may provide a more sustainable and environmental friendly waste management method compared to standard practices. In addition to the regulation of the moisture content, in cold regions air is sometimes injected to promote aerobic stabilization of the landfilled waste. Low temperatures can be a problem and aerobic composting stimulated by aeration can be used to heat up the bioreactor to the required mesophilic or thermophilic range during the starting phase (Reinhart and Townsend 1998). Nutrient level and pH have to be controlled as well. The pH affects the activity of methane forming bacteria. The range of 6.8 to 7.4 is known as the optimum pH for the methane forming bacteria. Nutrient addition is not common and is generally reviewed as not needed (Townsend et al. 2008).

The terminologies introduced here have been largely borrowed from Townsend et al. (2008).
Anaerobic Landfill Bioreactor: in an anaerobic bioreactor landfill, moisture is added to the waste mass in the form of recirculated leachate and other sources to obtain optimal moisture levels. Biodegradation occurs in the absence of oxygen (anaerobically) and produces landfill gas (LFG). Landfill gas, primarily methane and carbon dioxide can be captured to minimize greenhouse gas emissions and for energy generation.
Aerobic Landfill Bioreactor: similar to the anaerobic bioreactor, the aerobic landfill bioreactor has recirculation of leachate to adjust the moisture content. Air is injected into the

waste mass, using vertical or horizontal wells, to promote aerobic activity and accelerate waste stabilization.

Hybrid (Aerobic-Anaerobic) Landfill Bioreactor: the hybrid bioreactor accelerates waste degradation by employing sequential aerobic-anaerobic treatment to rapidly degrade organics in the upper sections of the landfill and collect gas from lower sections. Operation as a hybrid system results in the earlier onset of methanogenesis compared to aerobic landfills.

Semi-Aerobic Landfill Bioreactor: this is a landfill where natural ventilation via the leachate collection system promotes aerobic stabilization of the leachate.

As-Built Landfill Bioreactor: this type of LFB is conceived from the beginning (or near the beginning) as a bioreactor; the construction (and perhaps the operation) of the bioreactor components occurs while waste is actively deposited in the landfill. For as-built bioreactors more choices for selecting liquid addition techniques are available in comparison to retrofit bioreactors.

Retrofit Landfill Bioreactor: this type of LFB is not originally conceived as a bioreactor; the construction and operation of bioreactor components occurs after most or all of the waste has been placed. The methods that can be used for liquids addition are limited compared to as-built bioreactors.

In this chapter the focus is on anaerobic landfill bioreactor technology and its applicability in East Africa.

3.1.2 Landfill bioreactor potentials

Comparing anaerobic LFBs with sanitary landfills, LFBs have the following advantages: (i) Increased waste settlement rates enable utilization of the liberated airspace and hence increased the landfill capacity; (ii) Improved opportunities for in situ leachate treatment; (iii) More rapid LFG production and maximization of its capture which may improve the economics of gas recovery (Barlaz and Reinhart 2004) and abatement of greenhouse gases; (iv) LFBs also aim to minimize the landfill stabilization time which may lead to a reduced level of effort during the post-closure period and a shorter period of monitoring and liability retention. Accordingly, LFBs offer much potential as a viable waste disposal technology as summarized in Table 3-1 adopted from ITRC (2005).

Table 3-1: LFB primary and secondary advantages

Primary advantages	**Secondary advantages**
Stabilization of waste in a shorter time	Optimization of waste emplaced in a landfill
In-situ leachate treatment	Reduced leachate handling costs
Enhanced LFG generation rates	Potential for LFG to be a revenue stream
Reduced post closure care	Reduced air and leachate emissions
Efficient utilization of landfill capacity	Consistency with sustainable landfill design

3.1.3 Landfill bioreactor concerns

Landfill bioreactors have potential benefits, but they also raise concerns with regard to leachate seeps, slope stability, excessive temperatures, gas emissions and odour control. When the liquids are added at a high pressure or at a flow rate higher than the local infiltration rate or absorption capacity of the waste mass, there is a possibility of seeps. Furthermore, since liquids are added in the bioreactor, internal pore water pressures may increase and thus reduce the shear strength of the waste. Excessive pore water pressures can cause slope failures. High temperature occurring in aerobic bioreactor landfills is another concern because waste temperature may increase significantly and if not controlled, can cause fires. Another concern is gas and odorous emission which if not controlled leads to odours and other environmental problems.

3.2 Waste conversion processes in LFB

MSW placed in a landfill undergoes a number of simultaneous and interrelated biological, chemical and physical processes related to the conversion of the organic material and other components of the waste, leading to the production of LFG and leachate. Numerous studies such as those by Barlaz et al. (1989), Christensen and Kjeldsen (1989), Siegrist et al. (1993), Sponza and Ağdağ (2004) and Jiang et al. (2007) have been carried out on the anaerobic biodegradation process in the landfills. These studies have characterized the stabilization of waste starting with the filling of fresh MSW to well-decomposed waste in terms of an idealized sequence of five stages namely: initial adjustment, transition, acid formation, methane formation, and final maturation. Each stage is characterized by the quality and quantity of leachate and LFG produced. Virtually all MSW landfills undergo these five stages of stabilization. Operating a MSW landfill as a LFB has an effect only on the rate and on the duration of the stabilization stages but not on the sequence of the stages (Pohland and Alyousfi 1994; Reinhart and Townsend 1998; Kim and Pohland 2003). Thus, it is important to understand each of the stabilization stages individually. The idealized waste degradation process assumes that the waste is homogeneous and of constant age. A LFB in practice with highly variable age and composition of wastes may yield a different picture. In general, the chemical reaction for anaerobic decomposition of MSW can be written using the Buswell's (1952) equation (Tchobanoglous et al. 1993):

$$C_aH_bO_cN_d + \left(\frac{4a-b-2c+3d}{4}\right)H_2O \rightarrow \left(\frac{4a+b-2c-3d}{8}\right)CH_4 + \left(\frac{4a-b+2c+3d}{8}\right)CO_2 + dNH_3 \ldots(3\text{-}1)$$

3.2.1 Degradation of carbon compounds

The first stage of degradation is the initial adjustment (hydrolysis -aerobic decomposition) in which organic waste decomposes under aerobic conditions during and soon after placement of wastes in a landfill where both oxygen and nitrate are consumed and soluble sugars serve as the carbon source for microbial activity. The decomposition is dependent on the availability of oxygen from the air trapped within the landfill. Carbon dioxide and water are the main products with carbon dioxide released as gas or absorbed into water to form carbonic acid, which gives acidity to leachate. In well-run landfills this stage lasts only a few days or weeks.

The second stage (hydrolysis and fermentation - acetogenic decomposition) is a transition stage in which oxygen is depleted and anaerobic conditions begin to develop. This stage is dominated by anaerobic and facultative microbes that convert carbohydrates, proteins, lipids and cellulose to carbon dioxide, hydrogen, ammonia, glycerol and carboxylic acids (predominantly acetic acid). By the end of this stage, COD and volatile organic acids can be detected in the leachate. The leachate contains ammonia-nitrogen in a high concentration and the pH of the leachate starts to drop due to presence of organic acids (mainly acetic acid) and the effect of elevated carbon dioxide concentrations. This stage may last up to several months in well-run landfills and is permanent in not well-run landfills.

As hydrolysis is rate determining, it is the most important part in the biodegradation process of the solid substrate in landfills (Gholamifard et al. 2008). It is the transformation of complex particulate organic matter into simple monomer or dimer forms that can pass the bacterial cell membrane (Gawande et al. 2010b). The enzymatic hydrolysis of solids is a microorganism-mediated reaction whereby complex insoluble organic material is solubilized by enzymes excreted by hydrolytic microorganisms (Veeken et al. 2000; Gawande et al. 2010a). The rate of hydrolysis depends on several physicochemical factors. These factors include pH, temperature, composition and particle size of the substrate, diffusion, alkalinity, dissolved oxygen level and moisture content of the solid waste, the presence of methanogenic bacteria and adsorption of enzymes to particles (Veeken et al. 2000; He et al. 2005; Gawande et al. 2010b). Hydrolysis of solids controls the rate of methane production. Leachate recirculation affects the hydrolysis of the solid waste if the VFA in the leachate is lower than the VFA in the water pores of the waste mass.

The third stage characterized by acetogenesis and rising methanogenic decomposition is the acid stage, whereby acidogens/acid formers sometimes called non-methanogenic microorganisms convert the organic acids formed in the second stage to acetic acid, carbon dioxide (principal gas generated during this stage) and smaller amounts of hydrogen gas. In this stage the generation of small amounts of hydrogen creates conditions suitable for methanogenic activity. The acidic conditions, high concentrations of chloride, ammonia and phosphate ions increase the solubility of metal ions. The BOD, COD and the conductivity of the leachate rise significantly. This stage lasts over a period of a few weeks. The methane concentration in the gas stage begins to rise due to the carboxylic acids being utilized by methanogenic bacteria to produce methane, carbon dioxide and water.

The fourth stage is the methane fermentation stage whereby methanogenic and strictly anaerobic sulphate reducing bacteria convert the carboxylic acids and hydrogen gas formed in the third stage to methane and carbon dioxide. The conversion of the acids results in a rise of the pH within the landfill and the leachate to reach more neutral values which is the ideal condition for methanogenic microorganisms and consequently the reduction of the biochemical oxygen demand, the chemical oxygen demand and the conductivity of the leachate. Methanogenic bacteria are very sensitive to a drop in pH, so that the digestion of waste is obviously a delicate balance between the rate of hydrolysis, acidogenesis and methanogenesis (Chaggu 2004**).** This is the longest stage of waste degradation in landfills during which significant amounts of methane are generated.

The final stage is the maturation stage of waste degradation. The available biodegradable organic matter and the acids formed in the second stage are completely converted to methane and carbon dioxide in the fourth stage. In this final stage the gas production drops dramatically and leachate strength stays steady at much lower concentrations while aerobic conditions begin to develop and strict anaerobic microorganisms are replaced. The leachate will often contain humic and fulvic acids that are responsible for transport and behaviour of pollutants such as heavy metals and hydrophobic pollutants. Upon successful completion of this stage the remaining waste would be considered as biologically inert and no further emissions occur.

The characteristics of these sequential waste degradation stages are reflected in the quality of the landfill leachate. In the first stages, lasting about 8 weeks leachate COD_{tot}, VFA and ammonium (NH_4^+) concentrations rise to reach a ceiling after which in the following stages the concentrations gradually decrease (Kiely 1997, p.679). The highest values for the parameters COD_{tot} , VFA and NH_4^+ reached in a lab-scale reactor with municipal solid waste were about 100, 30 and 1.5 gr/l respectively. If landfill bioreactors with leachate recirculation are used the leachate concentrations of the mentioned parameters can be significantly lower (Sponza and Ağdağ 2004). In a large-scale landfill where waste is placed over a long period of time, the waste stabilization processes/stages tend to overlap and the leachate and gas characteristics reflect this phenomenon. The application of these stages to a MSW landfill setting is illustrated in Figures 3-1.

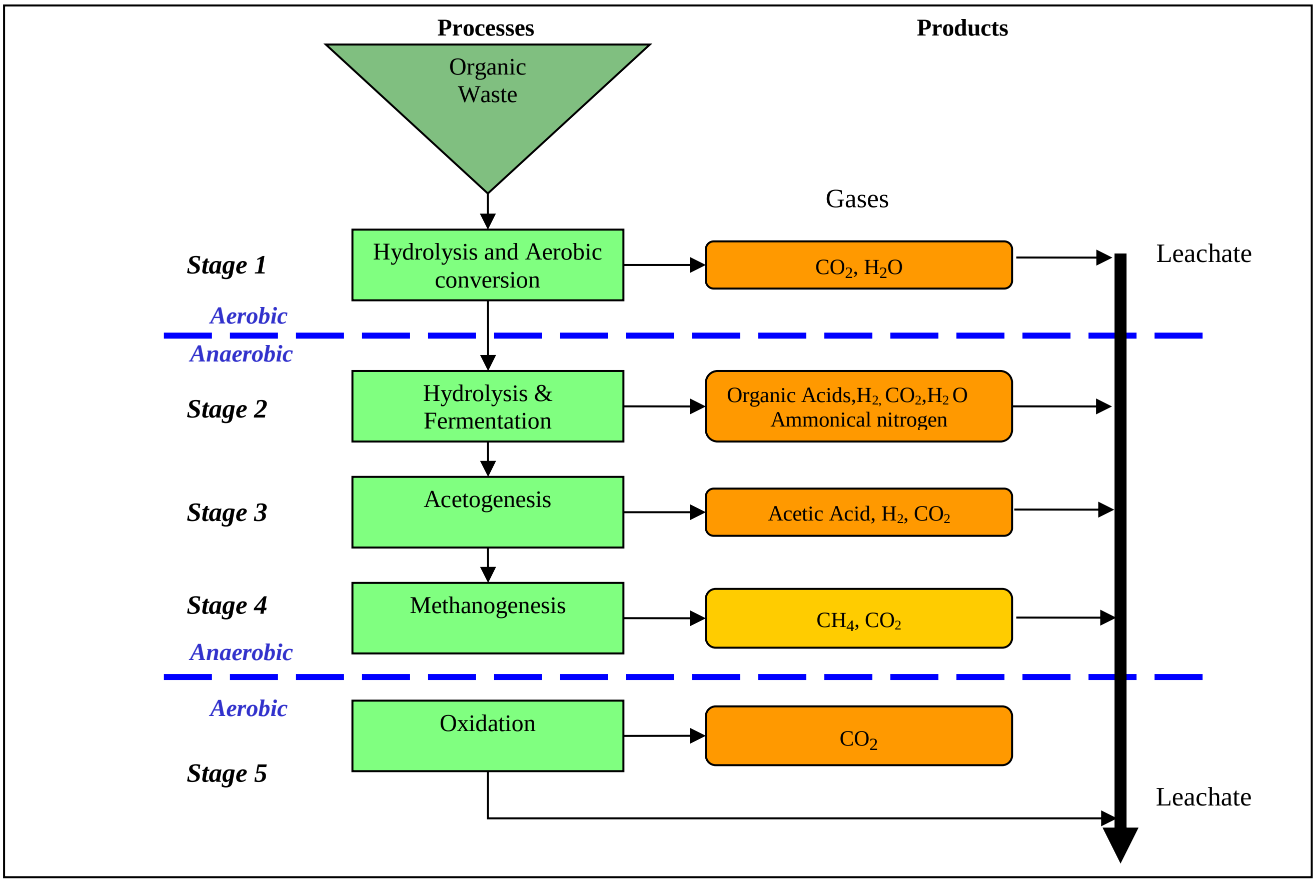


Figure 3-1: Major sequential stages of waste degradation in landfills
***Source**: Adopted from Waste Management Paper 26B, 1995 in Williams (2005)*

3.2.2 Conversion processes of nitrogen compounds

The nitrogen content of MSW is less than 1%, on a wet-weight basis (Tchobanoglous et al. 1993) and is composed primarily of the proteins contained in yard wastes, food wastes, and biosolids (Burton and Watson-Craik 1998). As the proteins are hydrolyzed and fermented by heterotrophic microorganisms, ammonia-nitrogen is produced by a process termed ammonification. In landfills, any ammonia produced may redissolve and react with organic matter before exiting the landfill (Berge et al. 2005).

Table 3-2 provides leachate ammonia-nitrogen concentration ranges for both conventional and bioreactor landfills as a function of waste age as summarized by Reinhart and Townsend (1998).

Table 3-2: Ammonia-nitrogen concentration ranges in leachate of conventional landfills and Landfill Bioreactors

Stabilization stage	Conventional landfills (N, mg/L)	Landfill bioreactors (N, mg/L)
Transition	120-125	76-125
Acidogenic	2-1030	0-1800
Methanogenic	6-430	32-1850
Maturation	6-430	420-580

Most of the ammonia-nitrogen in landfill leachate will be in the form of the ammonium ion (NH_4^+) because pH levels are generally less than 8.0 (Reinhart et al. 2002). As ammonia may be harmful to health and the environment and possibly also to the proper functioning of a LFB, an understanding of the fate of nitrogen in LFBs and possible mechanisms for ammonia-nitrogen removal is critical to both a successful and economic operation (Berge et al. 2005). Operating the landfill as a bioreactor provides opportunities for in situ nitrogen transformation and removal processes. The abatement of such ammonia pollution problems in leachate is discussed in detail in chapter 4. Here, the processes that determine the fate of ammonia in landfills are briefly described. These are nitrification, denitrification, sorption, volatilization, anaerobic ammonium oxidation (ANAMMOX), and nitrate reduction.

Nitrification can be a significant nitrogen removal pathway but in landfill environments nitrification is complicated by oxygen and temperature limitations, heterotrophic bacteria competition, and potential pH inhibition. Because nitrification is an aerobic process, it may be restricted to upper portions of the landfill or the cover where air may penetrate (Burton and Watson-Craik 1998). It is expected that in situ nitrification may increase in landfill since older waste contains fewer biodegradable organics, less competition with heterotrophs for oxygen will occur (Berge et al. 2005). pH may also be a complication during nitrification processes in landfills. Because nitrification consumes alkalinity, there may not be sufficient alkalinity present to buffer pH changes that would result from nitrification of high ammonia-nitrogen leachates. It is possible that alkalinity may need to be added to the landfill to buffer the leachate.

Denitrification is brought about by heterotrophic, facultative aerobic bacteria, that use nitrate as an electron acceptor when oxygen is absent or limiting. At the same time, these bacteria require a sufficient source of organic carbon for high nitrate removal rates. The advantage of denitrification is the simultaneous carbon and nitrate conversion without requiring oxygen input. There are studies that have evaluated in situ, or partially in situ, denitrification at both laboratory and field scale. Burton and Watson-Craik (1998) tested a landfill cell to denitrify externally nitrified leachate and results showed that the nitrate returned to the landfill cell was consumed under the anoxic landfill conditions. Price et al. (2003) conducted studies evaluating the ability of older waste to denitrify nitrified leachate and demonstrated that the landfill does have the capacity to denitrify, and that fresh waste contained enough organic carbon to support denitrification, while older waste required the addition of an external carbon source. Onay and Pohland (2001) observed the presence of autotrophic denitrification and concluded that autotrophic denitrification accounted for between 15% and 55% of the nitrate conversion to nitrogen gas, with the variation being attributed to the mass of organics present in the system. Vigneron et al. (2007) demonstrated that denitrification occurring during the acidogenic stage was predominantly heterotrophic, while autotrophic reactions prevailed during the methanogenic stage (Chen et al. 2009). The advantage of autotrophic denitrification is its conversion of nitrate to nitrogen gas in the absence of an organic carbon source and its utilization of inorganic sulphur compounds.

Ammonium flushing is leaching of ammonia-nitrogen from the waste controlled by the volume of water passed through the landfill, the nitrogen content of the waste, and the ammonia-nitrogen concentration in the bulk liquid. Reducing ammonia-nitrogen concentrations in a landfill by washout and dilution to acceptable levels requires the addition of large volumes of water. Flushing results in the removal of ammonia-nitrogen from landfills but the added large volumes of water must be treated externally. Furthermore, when operating the landfill as a bioreactor, leachate is recycled, and hence ammonia-nitrogen is continually reintroduced to the landfill while additional ammonia is solubilized into the leachate. It is very important to keep the ammonia-nitrogen level low to avoid inhibition of ammonia or nitrite oxidation (Kim et al. 2006).

Ammonium is known to sorb onto various inorganic and organic compounds. Sorption of ammonia-nitrogen to waste may be significant in LFBs because of the high ammonium concentrations present. Sorption therefore allows for temporary storage of ammonium prior to subsequent processes, such as nitrification and volatilization (Heavey 2003). Sorption is dependent on pH, temperature, ammonium concentration, and ionic strength of the bulk liquid. For ammonia to sorb to waste particles, it must be in the form of ammonium (NH_4^+). The conductivity of landfill leachate is generally high (approximately 7,000 μmho/cm) and thus may influence ammonium sorption (Berge et al. 2005). As ionic strength of the bulk liquid increases, sorption of ammonium tends to decrease (Heavey 2003). Ammonium desorption kinetics may be dependent on ammonium removal in the bulk liquid. As the ammonium concentration in the bulk liquid decreases, e.g. due to flushing or other removal processes, ammonium is likely to be desorbed from the waste to regain equilibrium (Heavey 2003).

Volatilization only occurs when free ammonia is present. In conventional landfills, ammonia makes up approximately 0.1 to 1.0% (dry volume basis) of landfill gas exiting the landfill (Tchobanoglous et al. 1993). Ammonia is not a greenhouse gas, but adverse health and environmental effects may result from exposure to the gas. At pH levels above 10.5 to 11.5, most ammonia-nitrogen present in solution is in the form of free ammonia gas (NH_3). As temperature increases, more of the ammonia is converted to free ammonia gas because of the temperature dependence of the acid dissociation constant (Berge et al. 2005).

Other nitrification/denitrification process are ANAMMOX and dissimilatory nitrate reduction to ammonium. Biological oxidation of ammonia-nitrogen may occur under anaerobic conditions but it is questionable whether or not the ANAMMOX microorganisms will be able to compete with denitrifiers for nitrate and nitrite within landfills (Berge et al. 2005) Dissimilatory nitrate reduction to ammonium in anaerobic or anoxic environments may also occur in landfills whereby ammonium is produced as a result of nitrate reduction. This pathway is generally favored when the microbes are electron acceptor (nitrate) limited in high organic carbon environments (Berge et al. 2005).

3.3 Effects of environmental factors in LFBs

This section discusses environmental factors that affect the degradation processes in landfill bioreactors. These factors are moisture content, pH, temperature, inhibitory influences and toxic components.

Moisture content

The moisture content in a LFB is determined by the initial waste moisture content and the incoming rainfall at the input side and the removed leachate, the water consumed in the formation of landfill gas and water vapor in the escaping gas at the output side. Previous experience and research indicate that the control of waste moisture content is the single most important factor in enhancing waste decomposition in landfills (Pohland 1975; Reinhart and Townsend 1998; Valencia et al. 2009). The control of moisture content in LFBs depends much on the initial moisture content in the waste. In industrialized countries the moisture available in municipal waste is usually not sufficient to meet the microbial requirements, so that design and operational modifications are needed to add liquids to the landfilled waste. In this situation the benefits of increased moisture content include spreading of microorganisms within the LFB, limiting oxygen transport from the atmosphere, facilitating exchange of nutrients, and dilution of inhibitors. Studies conducted by Bae et al. (1998), Jain et al. (2005), Khire and Mukherjee (2007) and Manfredi et al. (2009) reported that leachate recirculation can be used to increase the moisture content in a controlled reactor system and thus provide the distribution of nutrients and enzymes between methanogens and solid/liquids. In tropical countries MSW often is much wetter than in industrialized countries. The waste moisture here may be above field capacity (FC) so that the waste will spontaneously loose water; after deposition in a landfill a considerable amount of leachate will have to be drained. The field capacity (FC) referred herein is the internal water storage in the LFB quantified as the moisture content that the waste can “hold” under the influence of gravity (Townsend et al. 2008)

The stimulatory effect on biodegradation of maintaining a 40% moisture content and above has been proved by numerous studies such as Pohland and Harper (1986), Ress et al. (1998), Price et al. (2003), Berge et al. (2007), He et al. (2007), Berge et al. (2009) and Benbelkacem et al. (2010). The strong effect of moisture content was also seen in the correlations of total mass loss and moisture content in full-scale landfills (Baldwin et al. 1998). The moisture content and distribution also provides feedback on the effectiveness of the recirculation system and indicates how much liquid can still be added into the LFB.

pH and alkalinity

The pH in the waste mass has a profound influence on the combined processes of hydrolysis, acidification and methanogenesis. It is also an important indicator of the state of these processes. The optimum pH for methanogenic bacteria lies in the range of approximately 6.8 to 7.4 whereas a pH range of 5 – 6 is better for the growth of fermenting organisms. In the operation of a landfill bioreactor the methanogens must be able to convert the acid and hydrogen produced by hydrolysis and acidification. An insufficient activity of methanogens may lead to accumulation of hydrogen and acids resulting in extreme decrease of the pH and collapse of the process. This collapse is also called ensiling. As the methanogens are the most sensitive microorganisms the goal of bioreactor operation is to maintain the pH at a neutral level and sustain acetoclastic methanogenic organisms.

The pH in a LFB can be controlled by means of acid (HCl) or base (mixture of NaOH and KOH) addition to the leachate as it was the case in the study by Veeken et al. (2000). Findings from this study implied that the hydrolysis rate of biowaste depended on the pH value. According to the degradation model proposed by Veeken and Hamelers (2000), the accumulation of metabolic intermediary products such as volatile acids, not only hinders methanogenesis but also hydrolysis and acidogenesis. Therefore, a balance between acid production and acid consumption is essential for a stable anaerobic process, optimized methanogenesis and waste stabilization. Some of the operational techniques have been presented over the years such as two-stage process with separated acidogenic process and methanogenic reactors (Dinamarca et al. 2003; Cooney et al. 2007). The two-stage process in which hydrolysis/acidification and methanogenesis are taking place in different reactors has the following advantages: it maintains optional environmental conditions for each group of microorganisms and accelerates the waste conversion, thus increasing stability of the product by balancing the acidogenesis and methanogenesis.

A study carried out by San and Onay (2001) showed that a four times per week recirculation strategy maintaining at least 40% moisture content, with pH control provides the highest degree of stabilization. Dinamarca et al. (2003) studied the influence of pH on the anaerobic digestion of the organic fraction of the urban solid waste in a two-phase anaerobic reactor, and the results showed that a degradation of total suspended solids (TSS= 75%) and volatile suspended solid (VSS=85%) were obtained in the reactors operated at pH 7 and 8 in an operation time of 25 days (Dinamarca et al. 2003).

The alkalinity of the liquid in a landfill plays an important role in maintaining the balance between acid production and consumption. On one hand, the naturally generated bicarbonate alkalinity maintains a pH close to neutral inside landfill cells (Ağdağ and Sponza 2005). On the other hand, VFA alkalinity contributes to the buffering of H_2CO_3, but is transient since the VFA concentration varies and therefore cannot be consistently relied upon. Therefore, adequate alkalinity, or buffer capacity, is necessary to maintain a stable pH in a reactor for optimal biological activity. Accordingly alkalinity addition was used in numerous studies to neutralize the pH in the anaerobic treatment of MSW (San and Onay 2001; Warith 2002).

Temperature

Temperature is one of the major (key) variables influencing the digestion of waste in anaerobic reactors (Chaggu 2004). Many studies have proved that microbiological degradation rate increases with temperature until a certain maximum level is reached. Studies by Baldwin et al. (1998) and Ress et al. (1998) reported that the optimum temperature for methane production from domestic refuse in a conventional anaerobic digester is about 40^oC and that temperature control offers a potential means of manipulating the methane content of LFG. Temperature measurement can be used to monitor or to control liquid injection since good wetting of the waste mass seems to result in the most uniform temperature.

Stimulatory and inhibitory factors

The anaerobic ecosystem is considered to be rather sensitive to inhibitors. Researchers have reported many inhibitors of anaerobic degradation, e.g. oxygen, carbon-dioxide, hydrogen, proton activity, nitrates, sulphide, heavy metals and specific organic compounds (Christensen and Kjeldsen 1989). Cations, such as sodium, potassium, calcium, magnesium and ammonium, have been observed to stimulate anaerobic decomposition at low concentration while they inhibit at high concentrations. Price et al. (2003) in their study on nitrogen management in bioreactor landfills by chemicals addition observed the inhibition of methane formation in the presence of nitrate and indicated the bulk of the refuse was exposed to nitrate after a nitrate addition began. Other studies have reported that a high sulphate concentration can inhibit methane generation. It has been speculated that CO_2 acts as an inhibitor through the raising of the redox potential (Hansson 1982) and it is possible that it acts as an end product inhibitor during acetate and propionate degradation. In an LFB, the anaerobic degradation of wastes needs nutrients (Hettiarachchi et al. 2009) such as nitrogen and phosphorous because they are essential for microbial growth (Zhao et al. 2008). As the anaerobic ecosystem requires much less nitrogen and phosphorous than the aerobic system, well-mixed waste landfills will in general not be limited by nitrogen and phosphorous. Sometimes, however, the heterogeneity of a landfill may limit the nutrients' availability to microorganisms. Other micronutrients, e.g. chromium, cobalt, copper, iron, lead, nickel, magnesium, potassium, selenium, sulphur and zinc, have been reported to be present in most landfills. At very high levels, they can also inhibit the biological activities (Zhao et al. 2008) but at low concentrations they positively affect the growth of anaerobic bacteria resulting in low concentrations of COD and VFA in leachate samples.

Toxic components and their influence

Heavy metals are the most toxic contaminants in both landfill sites and landfill leachate. Heavy metal pollution can be one of the major environmental impacts of landfills. Changing environmental conditions at a landfill site (i.e. through leachate recirculation) can induce non-linear behavior and the sudden release of heavy metals at a problematic concentration level (Slack et al. 2005). For a long period after the deposition of MSW at a landfill, and also after the closure of a landfill, the leaching of heavy metals will continue (Reinhart and Townsend 1998; Long et al. 2009b; Long et al. 2009a). The highest heavy metal concentrations are observed during the acid formation phase of waste stabilization when the pH decreases to acidic values ($pH<7$) (Erses and Onay 2003; Long et al. 2009a). Therefore, continued attention to heavy metals after deposition of MSW is necessary (Øygard et al. 2008). It is known that heavy metal speciation in the environment is largely controlled by processes of precipitation, adsorption, and complexation. The distribution of heavy metals in the various phases determines their behavior in the environment, their mobility and bioavailability, while they can be removed from solution as sulphide precipitates if sufficient sulphur is available under reducing condition (Long et al. 2009a). Visvanthan et al. (2010) showed that the mobility of the metals from landfilled E-waste was to be intensified with the long term disposal or stabilization within landfills. This was caused by the solubility of respective sulfides, hydroxides, or other precipitates, as well as the degrees and modes of complexation with organic substances as reported by Bozkurt et al. (2000) in Visvanthan et al. (2010).

3.4 Steering the operation of LFBs

In this section waste pretreatment, co-digestion with other wastes, aeration, leachate management, LFG generation and extraction and reactor configurations are briefly discussed as means to improve and steer the operation of LFBs.

3.4.1 Waste pretreatment

Landfilling of properly pretreated wastes improves landfill behavior, characteristics, and operation. Pretreatment may consist of 1) mechanical disintegration (i.e. reduction in the size of the particles) bringing about an increase of the specific surface area; 2) biological pretreatment that promotes the hydrolysis of organic matter by enzymes or composting; 3) physico-chemical treatment by way of oxidative, chemical, thermal processes or a combination of thermal and chemical pretreatment. Whatever the pretreatment may be, the objectives are to obtain an extension and an acceleration of the stabilization process, an increased amount of LFG and a reduction of the digestion time (Mata-Alvarez 2002, p 202).

3.4.1.1 Mechanical disintegration

One way of improving performance of landfills operating as bioreactors is pretreatment of wastes by reduction of the particle size (Mata-Alvarez et al. 2000, p 201). Shredding and compaction are mechanical/physical waste pretreatment methods. Mechanical shredding can be efficient and effective in opening bags and reducing particle size. Relatively dry waste materials enclosed in plastic garbage bags do not break down even if wet conditions exist in the rest of the waste mass. In experiments on anaerobic thermophilic food waste digestion Kim et al. (2000) found that an increase of the average particle size from 1.02 mm to 2.14 mm led to a decrease of the maximum substrate utilization rate. The results revealed that particle size is one of the most important factors in anaerobic food waste digestion. A size reduction of the particles and the resulting enlargement of the available specific surface can support the biological processes and the effect being reduction of the digestion time (Palmowski and Muller 2000).

Warith (2002) also carried out a study to determine the effect of solid waste particle size, leachate recirculation and nutrient balance on the rate of MSW biodegradation. Larger particles of the collected waste were broken down to smaller size, and the material was thoroughly mixed prior to loading of the bioreactor cells. The MSW utilized in this experimental study was 60% organic matter and was shredded to a maximum of 150 – 250 mm in size. The study indicated that the smaller the size of the MSW the faster the biodegradation rate of the waste. It was also shown that the average pH levels of the shredded waste leachate samples were more neutral (7 - 8) than the un-shredded MSW leachate. In order to identify the effects of shredding and compaction of MSW Sponza and Ağdağ (2005) compared simulated LFBs loaded with shredded waste having a diameter of 50 – 100 mm, a reactor loaded with compacted waste and a control reactor loaded with raw waste . After 57 days of anaerobic incubation, it was observed that leachate of the reactor with shredded waste had a pH near neutral (i.e. 7.25) and COD and VFA concentrations lower than the control reactor and the compacted waste reactor. It was also found that MSWs having small particle size exhibited fast biodegradation. A BOD_5/COD ratio of 0.44 of the leachate achieved in the reactor with shredded waste indicated better MSW stabilization than in the reactor with compacted waste and the control reactor. However shredding is an intensive, high maintenance and high cost activity, which may not always be cost-effective. Shredding is mechanized, energy intensive and requires skilled labor thus not a suitable pretreatment option for developing countries.

3.4.1.2 Biological pretreatment

Composting

Composting (biological pretreatment) is a bio-oxidative process involving the mineralization and partial humification of the organic matter. Pretreatment of solid waste in a composting stage is proposed by Capela et al. (1999) in Mata-Alvarez (2002). The objective is to achieve pre-degradation of volatile solids and to decrease the inhibition of the methanogenic conversion due to acidification. As an illustration, the rate of volatile solids degradation of un-pretreated and pretreated substrate was 34% and 50% respectively after 49 days of anaerobic digestion. Furthermore, the degree of composting also enhanced the anaerobic digestion of wastes. A degree of composting of 10% provides a high rate (70%) of volatile solid reduction and a methane production of about 0.047 m^3 CH_4/kg VS. Therefore, with a low level composting, high methane production rates can be achieved (Mata-Alvarez 2002).

Aerobic thermophilic digestion

Another aerobic microbial process for pretreatment of waste is aerobic thermophilic digestion. This biological pretreatment process has the ability to produce a digested substrate that can be post-treated in mesophilic anaerobic digestion with enhanced volatile solid reduction and higher biogas production. Several authors reported by Mata-Alvarez (2002) mentioned contradicting findings. Hegaswa and Katsura (1999) found that organic sludge solubilized under slightly aerobic thermophilic conditions can generate 1.5 times as much biogas as untreated sludge. In contrast, Pagilla et al. (1996) showed that pretreated sludge produced biogas at a rate of 0.761 m^3/kg VSS which is lower than untreated sludge whose biogas production rate was 0.918 m^3/kg VSS. Whatever the possible strengths of this method, complete aerobic thermophilic digestion is expensive due to its large oxygen demand and it is capital intensive to build reactors from materials capable of minimizing heat loss (Mata-Alvarez 2002, p 211).

BIOCEL process

The BIOCEL process is an anaerobic digestion technology for organic fraction of municipal solid waste (OFMSW) based on a batch-wise digestion at high solid concentrations at mesophilic temperatures. Wastes are mixed with inoculum and then sealed into an unstirred batch reactor. The mixing of inoculum and the substrate is only carried out during the loading the BIOCEL reactor. When the pure organic fraction (without addition of inoculum) of MSW is anaerobically digested at 35% total solids (TS), acid formation starts within 2 hours (ten Brummeler et al. 1991). The anaerobic digestion is carried out in rectangular concrete digesters where waste is kept approximately 21 days until biogas production ceases. Subsequently, the digesters are unloaded. The floors of the digesters are perforated for leachate collection. The leachate collected during digestion process is recirculated back to the waste in the BIOCEL reactors.

For the BIOCEL-system it is essential to control emissions of odour and gases after opening and closing the doors of the digesters. Since biogas can make an explosive mixture with air, special equipment is installed and additional measures are taken (ten Brummeler 2000). All digesters have two special ports in the reactor cover where gases can be injected and drawn respectively from the digesters.

Procedures for opening and closing of the digester doors as well as odour control have been clearly described by ten Brummeler (2000). After the digester has been closed the headspace is still filled with air/21% oxygen. This oxygen might be used by facultative anaerobic micro-organisms while degrading organic matter. A potential amount of biogas would be lost. In order to prevent this loss, and to prevent the inhibitory action of oxygen towards the otherwise strict anaerobic methanogens, oxygen is flushed by CO_2-enriched off-gases. Likewise, after digestion is terminated, the headspace of the digester still contains methane rich biogas thus opening of a digester door would result in an explosive mixture with the incoming air. Therefore the headspace is flushed again with CO_2-enriched rich off-gases. The off-gases drawn from the digesters then require to be treated. Odour control in the BIOCEL-system is essential during unloading the biowaste from the trucks and opening the digesters to move the digested waste to the LFB. According to ten Brummeler (2000), the emissions of the digested biowaste mostly consist of ammonia and the pH is around 8.

3.4.1.3 Physico-chemical pretreatment

Oxidative physico-chemical pretreatment processes include wet oxidation and ozonation. Both these processes have been studied on solids/sludge in wastewater treatment. The basic principle is to enhance the contact between molecular oxygen or ozone and the organic matter to be oxidized after which the latter may be converted to methane more easily. The wet oxidative process works best at high temperature conditions to reduce total solids and volatile solids. In the lower temperature of 200^{o}C total solids and volatile solids destruction was 20 and 40% respectively, compared with 65 and 90% at 300^{o}C. The heat requirements to achieve high temperature renders this pretreatment option expensive. Ozonation also has its disadvantages such as a possible generation of toxic byproducts or high ozone emissions. Toxic compounds that originate from ozone reactions with some organic compounds are organic peroxides, low molecular-weight alcohol, some carboxylic acids and aldehydes (Mata-Alvarez 2002, p 212-213).

The use of acids or alkalis also falls in the category of physico-chemical pretreatment options with the latter being more compatible with the anaerobic digestion process. NaOH is commonly used for pretreatment of lignocellulosic materials because it provides better anaerobic digestion performance than KOH, $Mg(OH)_2$ and $Ca(OH)_2$. According to Mata-Alvarez (2002) studies conducted in the 1990s showed that low alkaline pretreatment improves the performance of subsequent anaerobic digestion. Using 20 meq NaOH/L of sludge for 24 hours at room temperature improves volatile solid removal in the range of 25 - 35%, COD removal of 30 - 75% and gas production of 29 - 115% compared to un-pretreated sewage sludge. Pretreatment with a dose of 1g NaOH/gVSS solubilized 15% VSS and methane production increased by 50% more than the untreated waste. In every case, potential toxicity problems must be considered in the chemical pretreatment of waste and particularly the inhibition or toxicity due to high ion concentrations (Mata-Alvarez 2002, p 216). The cost of the chemical pretreatment can be high and unaffordable by municipalities and cities of developing countries. Furthermore the residue as a result of chemical addition weighs heavily on the cost of disposal thus rendering pretreatment by physico-chemical pretreatment methods not a plausible option for developing countries like Tanzania, East Africa.

3.4.1.4 Thermal pretreatment

Thermal pretreatment is usually applied as a conditioning process to improve dewatering properties of raw or digested sludges. Mata-Alvarez (2002, p 218) summarized the advantages and disadvantages of this pretreatment option as: (i) hydrolysis of a large part of the particulate fraction; (ii) the production of VFAs which are easily converted to biogas in a subsequent biological step; (iii) easy hydrolysis of the remaining particulate fraction contained in the thermally pretreated waste-sludge by an anaerobic consortium. Major disadvantages are: (i) odour production; (ii) corrosion and fouling of heat exchange tubes; (iii) high energy requirement. The latter disadvantage is a stumbling block for such a pretreatment option especially in developing countries. Since this method is not applicable for MSW pretreatment then it is not further discussed in this chapter.

Summary

As shown in subsection 3.4.1 several pretreatment methods are proposed in literature. Among the methods mentioned here composting and the BIOCEL process (batch-wise anaerobic digestion) are the most feasible methods for developing countries. The BIOCEL process not only pretreats the waste but also reduces the volume of waste to be disposed and the bio-gasification of organic wastes in a short period of time is an advantage that is worth investing on compared to composting. The other processes are judged too complicated and expensive for developing countries.

3.4.2 Co-digestion

An interesting option for improving the biogas yields of anaerobic digestion of solid wastes is co-digestion with other wastes (Mata-Alvarez et al. 2000). At a right choice of the substrates the benefits of the co-digestion include: dilution of potential toxic compounds, improved balance of nutrients, synergistic effect of microorganisms, increased load of biodegradable organic matter and increase in biogas yield. Additional advantages include achievement of better handling, hygienic stabilization and increased digestion rate (Sosnowski et al. 2003) and process stability and economic feasibility. The key for co-digestion lies in balancing the macro and micronutrients, C:N ratio, pH, inhibitors/toxic compounds, biodegradable organic matter and the dry matter (Mata-Alvarez 2002, p 183-184). Co-disposal of MSW and sludge from municipal wastewater treatment plants has a significant effect upon the generation and quality of leachate (Ağdağ and Sponza 2007). Co-digestion of the organic fraction of municipal solid waste with other organic substrates, such as sewage sludge, livestock waste, industrial organic waste (e.g. waste from abattoir and meat-processing industries) etc., has shown several advantages. Several studies about co-digestion of MSW mixed with primary sewage sludge indicated the importance of utilizing reactors with a high solid content. Sosnowski et al. (2003) found that the anaerobic co-digestion of sewage sludge and OFMSW seemed to be an attractive method for environmental protection and energy savings. Similarly, a study by Warith (2002) demonstrated that the highest degree of settlement (about 50%) of a landfill was achieved through addition of sewage sludge. Ağdağ and Sponza (2007) showed that co-digestion of industrial sludge and OFMSW in anaerobic simulated landfilling reactors had a positive effect on COD and VFA reduction and pH adjustment.

Currently in Tanzania sewage is treated separately from other waste such as waste from abattoirs, industries and livestock waste. Opportunities for co-digestion have not been explored in developing countries but given a thorough feasibility study it can also be of value to MSW management in Tanzania and East Africa.

3.4.3 Gas extraction systems

LFG migrates primarily by convection and molecular diffusion into the atmosphere. LFG moves along routes characterized as the path of least resistance that will allow it to escape either by venting through the cover or by moving through the sides to the surrounding soil (El-Fadel et al. 1997) if uncontrolled. This migration pattern changes when gas recovery and/or control systems are introduced. Control systems can be passive or active (Tchobanoglous et al. 1993, p 402, p 406).

In passive control systems, the pressure of the gas generated serves as the driving force for its movement. Passive control systems include pressure relief vents, perimeter interceptor or barrier trenches and impermeable or sorptive barriers within the landfill. Such systems are only preferred during times when LFG is being produced at a high rate thus providing paths of lowest resistance for the gas flow in the desired direction.

In active gas control systems, energy in the form of an induced vacuum is used to control the flow of gas. Vertical and horizontal wells and sometimes their combination are commonly used for extraction of LFG. Typical vertical gas extraction wells are uniformly spaced in such a way that their spherical radii of influence overlap. Extraction well design (Tchobanoglous et al. 1993, p 411) consists of pipe casing usually PVC or PE. The wells are typically designed to penetrate to 80% of the depth of the waste in the landfill. Vertical wells are usually installed after the entire landfill or portions of the landfill have been completed.

Horizontal gas extraction wells are an alternative use of the vertical gas recovery wells. These wells are usually installed after two or more lifts have been completed. A trench is excavated in the waste matrix after which a perforated pipe with open joints is installed and backfilled halfway with gravel and capped with the waste. The backfill and capping is for the wells to withstand the landfill differential settlement expected to occur with the passage of time (Tchobanoglous et al. 1993, p 411).

The difficultly with gas control systems in LFBs is that they tend to fill with liquids as liquid and gas inside a landfill all follow the path of least resistance. If a gas collection device intercepts part of a saturated zone, liquids from this zone can migrate into the device. This problem has been observed with both vertical wells and horizontal trenches. The presence of moisture greatly reduces the ability of gas to move through the waste. If the waste surrounding a gas collection device is flooded, even if large amounts of gas are produced, gas will move elsewhere to a path with less resistance (Townsend et al. 2008).

Different techniques may be used to extract LFG. Possible options include use of a suction pump to induce vacuum in the vertical wells or horizontal trenches and to extract gas from the landfill interior. The vacuum has to be maintained in such a way so as not to draw air into the landfill, as the air drawn into the landfill may slow down the methanogenic microbial activity and can also result in landfill fires.

Biogas collection and energy recovery

Recovery of LFG for use as an energy resource is an area of vital interest since it is a creative solution for both environmental pollution and energy shortage. The generated LFG can be combusted directly in a modified natural gas or liquid propane combustion system or used to run internal combustion engines to generate electricity delivered to the national grid. Biogas can be purified to be equivalent to natural gas by scrubbing and removing water, carbon-dioxide and trace gas components. Table 3-3 adopted from (de Mes et al. 2003, p 80) provides an overview of techniques used for biogas treatment. The purified gas may be pumped into a natural gas pipeline as renewable natural gas or utilized directly.

An LFB generates more LFG in a much shorter time (i.e. 10 years) than a conventional landfill, so that early incorporation of LFG collection and management system are important in LFB design and construction. Degasification equipment such as flares, pipelines and blowers are gauged according to the most optimistic gas production estimates since it is necessary to assure the total environmental recovery of the LFB. For electricity generation, it is necessary to be aware of the best and worst case scenario for choosing the best type of generator for the installation (Zamorano et al. 2007). The economic viability of LFG utilization depends on a number of factors including the quality, local energy prices and choice of equipment and is based on cost-benefit analysis.

Table 3-3: Overview of biogas treatment techniques

Content removed	Principle	Technique
CO_2	Physical	Membrane separation
	Physical-chemical	Pressure swing adsorption Absorption techniques
H_2S	Physical	Membrane separation Molecular sleeves
	Physical-chemical	Absorption (to Fe_2O_3; with Caustic or Fe solution) Adsorption to Fe_2O_3 pellets Activated carbon filtration
	Chemical	$FeCl_3$ dosing to digester slurry
	Biological	Biological filtration Addition of air to a digestion process
Water/Dust	Physical	Demister Cyclone separator Moisture and water trap Cooling Absorption to Silica Glycol drying unit

3.4.4 Cell design and construction

The landfill consists of sections made up of cells and lifts provided with gas extraction and leachate recirculation pipes. The sections are gradually filled and later covered with a seal. Tchobanoglous et al. (1993) recommend each day's waste to form one cell and be covered with earth or any other suitable material. An emerging trend in sanitary landfill design, which bodes well with LFB evolution is to build deep cells. Deep cells improve compaction, and anaerobic conditions are more readily established. Furthermore, extremely deep (i.e. > 10 m) cells may be so dense in the lower portions such that permeability will inhibit leachate flow. Therefore the proposed LFBs in Chapter 7 are limited to 10 m height of cells.

3.5 Mass balances in LFB

As a result of waste conversion processes in an LFB, four significant impacts are expected. These include production of leachate and gas generation Other impacts are waste settlement and consolidation as well as stabilization of the waste which are discussed in section 3.6. Leachate production and treatment and gas generation are discussed here in brief and a detailed discussion can be found in chapters 4 and 5 respectively.

3.5.1 Leachate production and treatment

Leachate is the pollutant-laden liquid drained from the waste matrix (Mbuligwe and Kassenga 2007; Lozecznik et al. 2010). The quantities of leachate are to a high degree determined by the moisture content of the deposited waste and the incoming rainfall (see chapter 4). To avoid surface and subsurface water contamination, leachate collection systems are integrated in the landfill design and the collected leachate can be managed and treated on-site or off-site before discharge into the environment. In LFBs, the collected leachate is injected back into the landfill through a network of perforated pipes buried in the waste according to the moisture needs of optimized biodegradation. The management of leachate in LFBs is discussed in detail in chapter 4.

The generation of contaminated effluents (leachate) remains an inevitable consequence of LFBs requiring treatment. Factors affecting the quality of leachates are age, precipitation, seasonal weather variation, waste type and composition. The relation between the age of the landfill and the organic matter composition provides a useful criterion for the choice of a suited treatment process as noted by Kulikowska and Klimiuk (2008) that landfill age and ammonia-nitrogen concentration increase as organics concentration (COD) in leachate decreases. During the early stages of LFB operations, the leachate contains significant amounts of TDS, BOD and COD. The leachate needs to be pre-treated on site to meet the standards for its discharge into the municipal sewer or its direct disposal into receiving water bodies such as surface water. Reduction of quantity of leachate is considered a means of controlling the pollutant loading (Abbas et al. 2009). Leachate treatment is further discussed in chapter 4.

3.5.2 Landfill gas (LFG) generation and emission avoidance

In many of the issues related to LFG, determining its generation potential and rate is crucial as these are the most important parameters to size the gas collection and control system, the flaring system or the electric power plant. Many factors interfere in the generation of methane in a landfill, but the most important factors include the waste composition and the presence of readily degradable organic components, the moisture content, the age of the residue, the pH and temperature and the organic loading via the recirculated leachate (Jiang et al. 2007; Machado et al. 2009). Jiang et al. (2007) suggested that gas production was significantly enhanced in simulated bioreactor landfills as a result of both accelerated gas production rates and the return of organic materials in the leachate to the landfill for conversion to gas (as opposed to washout in conventional landfills) (Reinhart and Basel A. 1996).

According to Mehta et al. (2002) and Barlaz et al. (1990), the moisture content is a parameter that controls methane generation, since it stimulates microbial activity by providing better contact between soluble and insoluble substrates and microorganisms. As regards to the waste composition, different waste components will degrade at different rates over time. The rapidly biodegradable components normally include food waste and a portion (about 50%) of green waste (grass and leaves). The moderately biodegradable components include a portion of the paper waste and the remaining green waste, and the slowly biodegradable part includes the remaining portion of the paper waste (newsprint and coated paper), wood, textiles, and other materials. Plastic, glass, metal, concrete, rubble, and other inert materials are normally considered non-biodegradable.

VS is a good parameter to indicate the loss of organic material from a landfill over time (Mehta et al. 2002), however, alone it is not a good indicator of the remaining gas potential because not all the volatile material is converted into gas, as is the case of plastics and rubber (Machado et al. 2009). An estimate of the remaining biodegradable organic matter can be obtained by measuring the cellulose content of samples or correcting (reducing) the VS by the portion of non-degradable or recalcitrant matter. The same principle was used by

Tchobanoglous et al. (1993), who proposed using the lignin content to determine the biodegradable fraction of volatile solids.

LFG generation rates are also positively correlated with organic loading via the recirculated leachate. For example, Jiang et al. (2007) reported that when the influent concentration of COD increased from 3340 mg/L to 7810 mg/L, the LFG production rate increased from 29.1 to 910.4 L/week/ton waste (CH_4 50%, CO_2 50%, v/v), while it decreased to 35.6 L/week/ton waste when the influent COD dropped to 1590 mg/L.

It is important to control the generated LFG (Tchobanoglous et al. 1993) in order: (i) to reduce atmospheric emissions; (ii) to minimize release of odorous emissions; (iii) to minimize subsurface gas migration and; (iv) to allow for recovery of energy from methane. Reduction of LFG emissions, worldwide one of the most significant anthropogenic sources of greenhouse gas emissions, is an important issue in the Kyoto protocol.

As discussed above waste composition and the presence of readily degradable organic components, the moisture content, pH and alkalinity, temperature, the availability of nutrients and microbes, and the presence of inhibitors such as oxygen, heavy metals, and sulfates are all factors that influence LFG generation. Figure 3-2 adopted from El-Fadel (1997) depicts these variables as function of landfill operational practices.

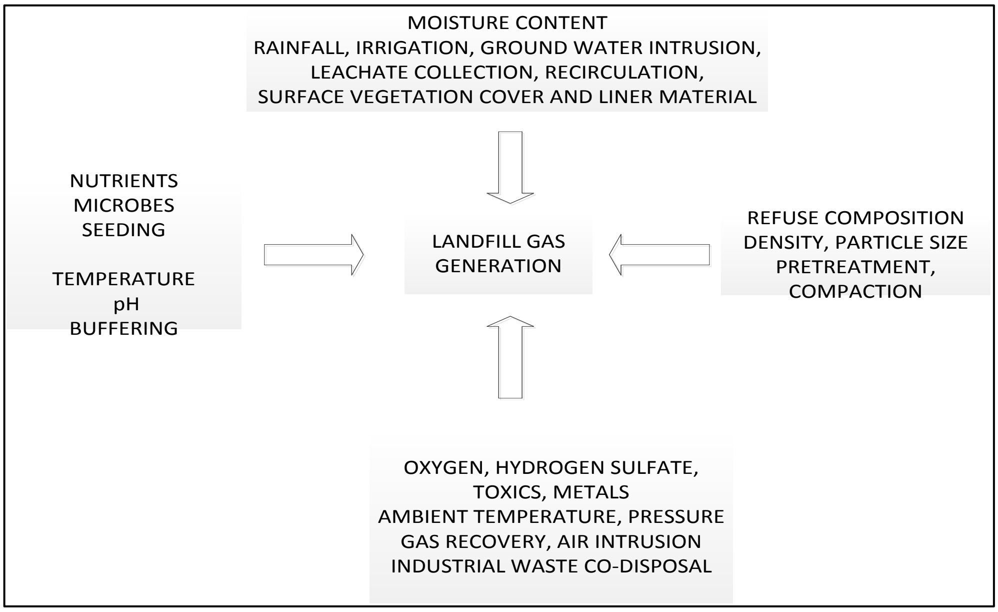


Figure 3-2: Landfill management factors influencing LFG generation.

Since landfill bioreactors produce more gas over a shorter period as shown in Figure 3-3 (Townsend et al. 2008) utilizing their gas for heating or electricity generation is economically more feasible than at regular sanitary landfills.

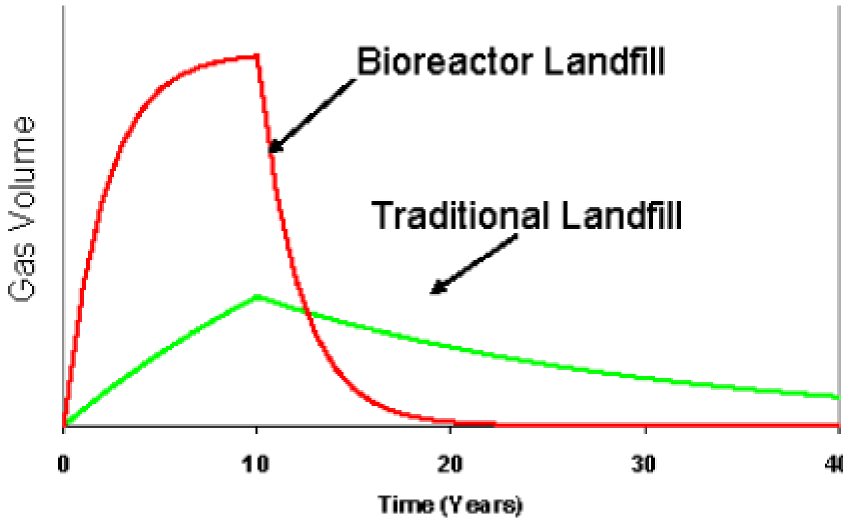


Figure 3-3: LFG generation in LFB versus regular sanitary landfill

It should also be noted that LFG has a heating value of approximately half that of natural gas and can in certain applications be used in place of conventional fossil fuels. In order to realize the advantages of landfill bioreactors the minimization of gas losses to the atmosphere is a prerequisite. The technical details of gas extraction are discussed in the next section.

Landfills are not point sources, but a diffuse source of methane. Moreover, the emission has a high temporal and spatial variability. Therefore, it is not easy to measure methane emissions. In the framework of abatement of greenhouse gases it is important to consider the sources and fate of carbon in landfills. In the context of this thesis, the carbon in deposited MSW is considered of biogenic origin meaning that it does not contain fossil carbon (USEPA 2002). A part of the deposited carbon remains in the landfill which may be considered as a man-made carbon sink. The remaining biogenic carbon is converted to methane and carbon-dioxide. Between these two only the methane escaping from the landfill to the atmosphere is assumed to contribute to global warming.

Figure 3-4 schematically illustrates the flows of carbon in a landfill. The generation of landfill gas and its global warming potential are elaborated in detail in chapter 7. In the next subsection practical measures to maximize landfill gas capture are discussed.

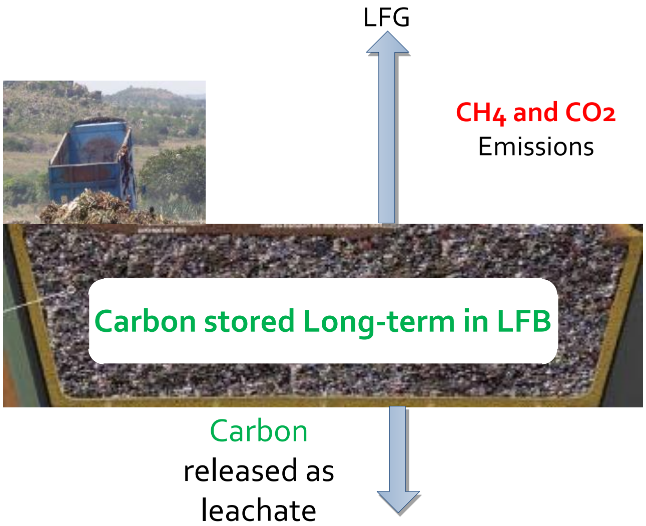


Figure 3-4: Carbon flows within LFB

3.6 Stabilization and settlement of the waste

As waste stabilizes at a landfill settlement or reduction of volume occurs and this settlement is a clear indication of the degree of waste stabilization, and hence the operation of the bioreactor. Different studies have viewed causes of MSW settlement in LFBs in different ways. Reinhart and Townsend (1998) classified the primary causes for settlement into four: reduction in void space and compression of loose material due to overburden weight; volume changes due to biological and chemical reactions and dissolution of waste matter by leachate; movement of smaller particles into larger voids; and settlement of underlying soils. El-Fadel and Khoury (2000) report five mechanisms governing settlement. They are classified as mechanical change, raveling, physicochemical change, biochemical decay, and interaction among these mechanisms and are briefly described in Table 3-4 adopted from (El-Fadel and Khoury 2000).

Table 3-4: Mechanisms of solid waste settlement

Mechanism	Description
Mechanical	Distortion, bending, crushing and reorientation of the materials; similar to the compression of non-organic soils
Raveling	Shifting of fine materials into the voids between larger particles
Physicochemical	Corrosion, oxidation, and/or combustion of the waste material processes
Biological processes	Aerobic/anaerobic decay of the waste material
Interaction	Above mechanisms could interact to cause additional settlement

According to Elagroudy et al (2008), a landfill is an interacting system of multiphase media (gas, liquid, and solid) with each phase exhibiting spatial and temporal variations. Therefore, MSW settlement depends on contributions from all three phases. Settlement is also known to be a function of many factors such as the material and the thickness of the cover, MSW composition like moisture and volatiles, density achieved after compaction of the landfill, self-weight, overburden, climate, method of filling, mode of operation, and more (Swati and Joseph 2008). Hettiarachchi et al. (2009), reports that there are two broad mechanisms that can be used to describe the settlement: mechanical compression and biodegradation-induced settlements. The rearrangement of MSW after biodegradation produces additional settlement. Thus the total settlement is a combined process of mechanical compression and biodegradation-induced settlement. All mechanisms of MSW settlement described by different authors can be classified into three main sequential phases as identified in the literature by El-Fadel (2000) in Elagroudy et al. (2008) and Swati and Joseph (2008). The sequential settlements are namely initial/immediate and rapid due to overburden pressure, primary settlement due to dissipation of pore water and void gases and secondary settlement due to creep of refuse skeleton and biological decay. Initial compression is rapid settlement that occurs instantaneously when an external load of mainly heavy overlying cover layers is applied, which may be very significant if the waste is not well compacted. Primary compression is mainly due to factors operative inside the waste matrix and marginally due to the continuing external stress factors. It is associated with the immediate compression caused by increase in voids ratio due to solids loss, pore water and gas as a result of superimposed loads. Secondary compression is caused by movement of the waste as a result of the continued decomposition for a long time until the waste is fully stabilized and it occurs over many years. A good correlation exists between settlement and organic destruction during waste degradation in MSW. A correlation factor of 0.885 between settlement and VS reduction and 0.635 between settlement and leachate dissolved organic carbon reduction has been reported by Swati and Joseph (2008).

In an LFB there is a much faster degradation of the waste than in sanitary landfills due to the recirculation of leachate. As leachate recirculation affects waste stabilization it also influences waste settlement. According to Hao et al. (2008) the MSW settlement ratio (settling height/initial height) in one of simulated landfill bioreactors reached up to 14%, while only to 0.52% in another reactor at the end of experiment. The difference in the settlement ratio is mainly because of higher biodegradation rate due to leachate recirculation in the reactor that achieved more settlement than in the reactor where only water was added. The difference in the settlement ratio between these reactors could be explained as follows. In the study by Hao et al. (2008), the operational procedure of leachate recirculation consisted of two steps: leachate discharge from the bottom and then leachate injection from the top inlet of the simulated reactor operated separately. Apparently, the first step caused increase in the void ratio of MSW and weakened its structural strength. The second step caused downward pressure on MSW in the reactor. The more leachate recirculated, the higher MSW settlement generated (Hao et al. 2008). After stabilization the remaining waste mass has characteristics similar to a low grade lignite or peat.

The opportunity to add more waste into the liberated landfill airspace extends the working life span of the landfill and defers capital and financing costs needed to locate and construct a new landfill resulting in capital savings and realized waste disposal revenues (Hettiarachchi et al. 2009; Elagroudy et al. 2008). It is also important to evaluate the impact of settlement on landfill components such as leachate recirculation systems and gas collection pipe networks (Hettiarachchi 2005) in (Hettiarachchi et al. 2009).

3.7 Closure stage of LFB

Closure of an LFB as considered in this chapter implies the site is no longer receiving fresh waste, all the cells are completely closed, there is no more LFG collection and no leachate recirculation. In order to evaluate the post-closure needs and/or to enhance the bioreactor operation of the landfill, information about the conditions inside the landfill body is required. Monitoring wells placed in the landfill body can be used to characterize the leachate quality which reflects the degradation stage of the waste as well as water movement in the landfill. (Sormunen et al. 2008).

3.7.1 Post closure care of LFB

Post-closure care (PCC) at a MSW landfill ensures that a solid waste facility is managed after final closure so that it does not pose a threat to human health and the environment. Post-closure care comes with certain costs. Long term post-closure maintenance and monitoring of landfills maybe financially unacceptable. It therefore becomes incumbent on landfill operators to ensure that the rate of degradation of waste is optimized in order to reduce the time-scale of their liability (Allen 2001). Reductions in post-closure care periods in comparison to conventional sanitary landfills have been cited as a potential benefit associated with LFBs. If the post-closure care period is reduced by 20 years, the post-closure care costs decrease by 25–30%, which corresponds to a 2% decrease in the total landfill costs (Berge et al. 2009). Thus, while reductions in post-closure periods appear to have a minor impact on overall project economics, they do represent significant savings over traditional landfills post-closure care costs. PCC performance is based on a systematic and hierarchical evaluation of (1) leachate, (2) landfill gas, (3) groundwater, and (4) the final cap.

3.7.2 Maintenance, monitoring and evaluation

Typical maintenance activities during the post-closure period include care of the following services:

- groundwater monitoring and annual reporting
- risk assessments, alternative concentration limits, and corrective action for groundwater
- landfill gas monitoring and corrective actions
- cover maintenance
- cap and storm water systems repair
- regulatory compliance.

USEPA (1998) details the PCC operation and maintenance requirements for the four systems that prevent or monitor releases from the landfill unit:

- leachate collection system
- landfill gas monitoring system
- groundwater monitoring system
- cover system

These operation and maintenance requirements are briefly summarized here.

Leachate collection and recovery system:

The purpose of a leachate collection system is to effectively collect and remove leachate from the LFB throughout its active and post-closure life. Routine monitoring and maintenance activities include maintaining and repairing leachate removal and transmission system elements (pump stations, meters, valves, manholes, transmission pipes, etc.), inspecting and maintaining leachate collection and storage systems, and sampling and analyzing leachate. Monitoring data such as leachate generation rates, the composition of the leachate and proximity to surface water, wetlands, and groundwater should be used to demonstrate that there is no uncontrolled leachate present at the site, that discontinuation of the leachate collection system is not a threat to human health and the environment, and that water quality standards in receiving surface water or groundwater are not violated.

Landfill gas monitoring system

Monitoring LFG is necessary at the LFB boundary and in buildings on site to verify operation and maintenance of the landfill gas extraction system. An adequate gas monitoring plan/network must be in place for a sufficient period of time to allow the migration of gas to be evaluated in order to address upgrades or repairs to LFG management system components, and mitigation of off-site gas migration concerns.

Groundwater monitoring system

The groundwater monitoring system has to be designed to allow collection of representative samples of groundwater for evaluating the potential for groundwater quality. Typically, the results from these monitoring events are compared to background conditions or health-based standards to demonstrate compliance or to establish trends that can be used later in the performance evaluation of different LFB elements, including determining an appropriate duration of post-closure care.

Cover system

Post-closure care includes maintaining the integrity and effectiveness of the final cover system, including making repairs to the cover as necessary to correct the effects of settlement, erosion, or other events, and preventing run-on and runoff from eroding or otherwise damaging the final cover.

3.8 Summary

Landfill is an essential part of an integrated waste management strategy, without which effective municipal solid waste management will not be possible in developing countries. The development of a truly sustainable landfill is important to the safe and effective management and control of municipal solid waste in the future.

What can be done is to significantly improve the open/controlled dumps to adoption of engineering techniques. Movement from the controlled dumping to engineered landfills may be viewed as costly but depends on the availability of physical and financial resources. Whether developing countries are prepared to pay in the short term the price for truly sustainable landfill development remains to be seen whereas the long-term benefits cannot be questioned.

The waste degradation in dumpsites and conventional landfills can be enhanced by operating them as an anaerobic bioreactor and eventually the stabilized waste mass with LFG and less polluted leachate that can be recovered creates valuable landfill airspace within a reasonable time scale. The underlying principle of the landfill bioreactor is that by optimizing operational control and environmental conditions within the waste particularly moisture content by way of recirculation of leachate, more rapid and complete biodegradation of municipal solid waste may be achieved and more biogas can be produced

As compared to industrialized-developed countries, the concept of landfill bioreactors technology is relatively very new to developing countries like Tanzania in East Africa. Table 3-5 is a summary of the various aspects that surround the introduction of LFB technology to East Africa and their respective descriptions which also include benefits that can be accrued by implementation. The aspects include to operate the LFB in anaerobic mode with the crucial benefit of biogas production as a result of biodegradation of organic matter which is the major component in the MSW generated in East Africa. Another aspect is the introduction of pretreatment of MSW in a BIOCEL-system. The BIOCEL-system is a proven technology with capability of biogas production from the rapidly biodegradables in MSW in a short time (about 20 days) and also waste volume reduction as a result of the rapid biodegradation. Other aspects include modalities of cell operation (filling time, length of active period, depth), leachate and gas collection systems, leachate recirculation and in-situ treatment. Included in the table is also the benefits that can be realized in comparison with existing landfill operations currently practiced in East Africa. The benefits are such as the use of LFB enhances stabilization of waste in a shorter time, efficient utilization of landfill capacity, more and rapid LFG (biogas) production, greenhouse gas emissions avoidance, control of odour, reduced leachate treatment costs and reduced post closure care of the landfill.

Table 3-5: Basic information and benefits for choice of applicability of LFB in East Africa

Aspects	**Description**
Feasible MSW Treatment Technology	Anaerobic treatment
Waste pretreatment technology • BIOCEL-system	Substantial amount of biogas production at short time
	Less volume of waste for final treatment
	No loss of biogas
Waste treatment technology • Standard landfill bioreactor (LFB)	Enhanced stabilization in a shorter time (10 years)
	Efficient utilization of landfill site
	Reduction of post closure care
LFB • Layout • Operation	>10 m waste height in a cell
	1 cell filled per week
	During 5 years the cell is fully active (leachate recirculation, gas collection)
	After 5 years cell is partially active (no leachate recirculation, gas collection)
	After 10 years cell is completely closed (no leachate collection, no gas collection)
Landfill gas (biogas)	Active gas production and collection
	GHG Emission avoidance
Leachate management	Collection of all produced leachate
	Vertical wells leachate recirculation system
	In situ leachate treatment via recirculation
	Reduction of leachate treatment costs
	Ex-situ leachate treatment

CHAPTER 4

Landfill leachate management

4.1 Introduction

This chapter focuses on the management of leachate emanating from the Landfill Bioreactor (LFB). Often reference is made to the sanitary landfill because most data about MSW and leachate management in the literature refer to sanitary landfills. Much less has been published about the specific leachate quality, quantity and treatment related to LFBs. A fundamental difference between the two landfill types, i.e. LFBs and sanitary landfills, is the recirculation of leachate in the LFB.

Leachate recirculation in a landfill may be applied for different reasons. Firstly, it may enhance the biodegradation and treat the leachate. Secondly, it may be applied in a way that only the first two steps of the biodegradation process, i.e. hydrolysis and acidification, are stimulated after which the acidified leachate may be treated in an anaerobic secondary reactor. Thirdly, leachate recirculation may be applied for de-nitrification of nitrified effluent of a leachate treatment system. Here, the carbon in the landfill serves as an electron-donor.

Generation of leachate and accordingly the treatment of leachate remains an inevitable consequence of the existing landfilling practice and this also holds for LFBs. This is particularly true where the landfilled wastes are relatively wet as is the case in Tanzania. The generated leachate needs to be treated to meet the standards for its discharge into municipal sewers or direct disposal into surface water. For design and operation of leachate treatment plants it is important to know the quantities and quality of leachate that requires management. The main issues to be addressed in this chapter are leachate production, leachate recirculation, characteristics of the leachate, technical aspects of leachate recirculation and proposed leachate treatment options for leachate from LFB.

4.2 Leachate production in LFB

In an LFB fresh or pretreated MSW is deposited in cells. The height of the cells gradually increases until a maximum is reached and the cell is covered with capping material and the leachate collection and recirculation systems are installed. During waste stabilization in the landfill cells the waste mass decreases. Firstly, a part of the waste is converted to biogas. During biogas formation a small part of the water in the waste may be consumed as reaction water and some water leaves the landfill as vapor in the biogas. Secondly, water in the waste is released as leachate. Leachate is generated by decomposition of material that contains moisture and the pressure of the waste mass on underlying layers of waste in a cell. The pressure squeezes leachate in excess of the field capacity out of the waste. Thirdly, there is rainfall that may penetrate into the cell. This adds to the additional leachate production. Fourthly, there is evaporation of water due to high temperatures experienced in tropical countries such as Tanzania and East Africa at large. Furthermore, a part of the waste mass leaving the cell consists of various types of solids contained in the leachate.

Some researchers have tried to develop models for predicting leachate quantity from landfills. The most frequently used model is the Hydrological Evaluation of Landfill Performance (HELP). The HELP model is useful for long-term prediction of leachate quantity and comparison of various design alternatives (Reinhart and Townsend 1998). Another model is the Deterministic Multiple Linear Reservoir Model (DMLRM) and the Stochastic Multiple Linear Reservoir Model (SMLRM). These models were developed to better simulate leachate generation at active landfills but not landfill bioreactors thus they are not going to be discussed in this thesis.

Another model for leachate calculation is presented by Tchobanoglous et al. (1993) in chapter 11. This model for sanitary landfills revolves around the incoming water (waste on the landfill and rainfall) and the dynamic behavior of the field capacity (FC) of the deposited waste. In the mentioned model the field capacity (i.e. the internal water storage in the LFB quantified as the moisture content that the waste can "hold" under the influence of gravity) is the key parameter in the calculation of the water balance in the landfill at any moment. The field capacity (FC) of the waste is calculated by means of the empirical equation:

$$\mathrm{FC} = 0.6 - 0.55 * \left(\frac{\mathrm{W}}{5400+\mathrm{W}}\right) \ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots \quad (4-1)$$

Here,
W is the weight (overburden weight) of the waste at the mid height of a cell/lift (kg/m^2)

This equation shows that FC decreases with increasing value of weight. The weight (W) increases with increasing landfill height and more leachate is produced with increasing pressure on the waste in the landfill.

The overall amount of leachate produced per unit area of a landfill depends on the following main factors:

- The height of the landfill cell/lift - the higher the cell/lift the more pressure on the waste below (m);
- Density of the waste - the higher the density the higher the pressure on the waste below;
- The dry weight of the waste per unit area;
- The weight of the cover - the higher the cover weight the lower the field capacity. This weight depends on the mass of cover material/per mass of waste deposited;
- The initial moisture content of the waste;
- A minor factor is the gas production. Gas production means a decrease of the weight in the landfill and therefore decrease of the value of W. Also the gas production implies a loss of water from the landfill as vapor and reaction water.

Putting the terms that compose the water balance into an equation (Tchobanoglous et al. 1993, p 424) equation (4-2) is obtained:

$$\Delta S_{SW} = W_{SW} + W_{CM} + W_{A(R)} - W_{LG} - W_{WV} - W_{E} - W_{B(L)} \ldots\ldots\ldots\ldots\ldots\ldots \quad (4-2)$$

Here:

ΔS_{SW} change in amount of water stored in landfilled waste (kg/m^3)
W_{SW} water (moisture) in the incoming waste (kg/m^3)
W_{CM} water (moisture) in cover material (kg/m^3)
$W_{A(R)}$ water from upper landfill layer (such as rainfall) (kg/m^3)
W_{LG} water lost in the formation of LFG (kg/m^3)
W_{WV} water lost as saturated water vapor in LFG (kg/m^3)
W_{E} water lost due to surface evaporation (kg/m^2)
$W_{B(L)}$ water leaving from bottom element (i.e. leachate) (kg/m^2)

Through equation (4-2), leachate production is the difference between the amount of water initially present in the waste and the cover (W_{SW} + W_{CM}) plus the water entering the waste in the landfill ($W_{A(R)}$) and the amount of water remaining in waste (S_{SW}) and released in gaseous form during the operation time of a landfill (W_{LG} + W_{WV} + W_E). In East African cities the collected waste is rather wet so that a significant amount of leachate is being produced at the transfer stations before the waste is transported to the landfill site.

The above mentioned factors and equation were established for sanitary landfills but also hold for LFBs, though with a slight difference. As leachate is recirculated a steady state with regard to the moisture content in the waste is reached sooner after deposition of fresh waste than in a sanitary landfill. The recirculating leachate fills the waste until or above field capacity, but whether this occurs depends on the initial moisture content and the rate of recirculation. Benson et al. (2007) remark that none of the full-scale landfills in the USA they had monitored appeared to have reached field capacity of 40 - 50%. This was probably due to their application of a low recirculation rate in the range of 21 to 163 l/m^2/year and also probably because the initial moisture content of the waste was low. Therefore, the design of the leachate management system for LFBs must take into account the initial moisture content of the waste and the access of rain to the waste mass.

In East African cities the waste deposited at landfills has a water content higher than field capacity (> 50%) (chapter 2). It may be expected therefore that immediately after deposition of the waste relatively large amounts of leachate are produced.

4.3 Leachate recirculation

4.3.1 Hydrodynamic behavior

A recirculated landfill can be considered as permeable bed. The maximum vertical rate of flow (m^3/m^2.d) occurs when the void volume is completely filled with liquid and determined by the packing and porosity of the bed, size of the materials in the bed and viscosity of the liquid (Darcy's Law, Carman-Kozeny equation). The maximum flow rate increases with porosity of the bed and size of the materials. Specific density of the waste and cover materials and landfill height also play a role. With increasing height and specific densities the pressure on lower layers will increase and the porosity decrease, thus lowering the maximum flow rate.

If the recirculation rate exceeds the maximum flow rate of a cell, flooding will occur. At too low flow rates the retention time in the landfill will be too long and recirculated leachate takes a very long time to reach the bottom of the landfill. In this case recirculation will have little influence on the processes in the landfill. This is the case in landfill bioreactors with a very low recirculation rate. From the above it can be concluded that the recirculation rate can be varied but should not exceed a certain maximum at which flooding would occur. The discussion about recirculation rates is elaborated on below in sub-section 4.3.2.

In literature leachate recirculation rates have been quantified in several ways. For landfills the recirculation rate is sometimes expressed as a surface loading rate (l/m^2.day or l/m^2.year). As quantification should say something representative about the influence of the recirculated liquid on the waste mass, another way of quantification proposed here is mass or volume of liquid per mass of wastes per unit time: e.g. l/ton per unit time. This is the mass-related loading rate. In this case the waste mass per square metre of the landfill has to be calculated to be able to convert given rates to l/ton.day. Especially in lab-scale studies where the waste depth is usually small mass-related loading rates are relatively high. Where necessary recirculation rates in this thesis will be expressed in both units.

4.3.2 Effects of recirculation

As indicated above there are several motives for landfill recirculation. Several studies have been reported on the question of landfill bioreactor behavior as function of recirculation rate. One could search for the rate for optimum biodegradation and in-situ leachate treatment or for the rate to obtain maximum acidification of the recirculated leachate. Here, a few articles about leachate recirculation rates are briefly reviewed. We focus on publications about the treatment of wet waste comparable to the waste in Tanzania.

Experiments by Sponza and Ağdağ (2004) showed that there is an optimum recirculation rate for waste biodegradation in a landfill. In lab-scale experiments these authors compared three anaerobic LFBs filled with kitchen waste and with leachate recirculation rates of 0, 9 and 21 l/day which corresponds to 0, 310 and 880 l/ton wet waste.day (0, 129 and 300 l/m^2.day). The reactor height was 100 cm. The waste they used had a high organic matter and moisture content and was inoculated with anaerobic sludge. At the recirculation rate of 310 l/ton.day they found an enhancement of biodegradation and concomitant lower COD and VFA concentrations in the leachate as compared to the single pass reactor. Here, recirculation improved biodegradation. At the higher recirculation rate of 880 l/ton.day, however, the conditions of biodegradation had deteriorated, so that leachate COD and VFA were higher than when the recirculation rate was 310 l/ton.day. At this high recirculation rate an accumulation of VFA to concentrations similar to the single pass reactor (no recirculation) had occurred which testified to an unbalance between acidification and methanogenesis. Also the biogas formation showed the highest values at the recirculation rate of 310 l/ton.day. It was evident that under the conditions of these experiments there was an optimum recirculation rate for methane production. The initial improvement of the biodegradation was attributed to spreading of methanogenic bacteria and enzymes from the added seeding sludge to the wastes, but a too high recirculation would lower the buffer capacity and increase the accumulation of VFA. Knowing the conditions for obtaining a highly acidified effluent is interesting when the aim is to design a two-stage reactor system with biomethanation in the second stage. The authors state that high recirculation rates would remove the activity of methanogens. It should also be noted that Sponza and Ağdağ (2004) used anaerobic seeding sludge in their experiments which usually is not always the case in normal landfill bioreactors. It is not clear what the effect of recirculation would have been if this seed sludge would not have been added.

Jiang et al (2007) varied the recirculation rate in four 5.7 m high pilot-scale reactors filled with 28 ton of wet commingled urban waste (density 0.93 ton/m^3) from 0 to 8 l/ton.day (0 – 30 l/m^2.day). The temperature varied between about 15 and 35 ^oC. They found that the generated LFG volumes increased with increasing recirculation rates. In all the reactors the pH remained above 6.1 helped by the addition of $Ca(OH)_2$ to the fresh waste, so that the initial leachate pH remained neutral to slightly alkaline. VFA concentrations in the leachate were initially high (4 -5 g/l) in all reactors but drastically decreased to below 500 mg/l after about 30 weeks accompanied by a strong acceleration of the LFG generation. This decrease occurred first in the reactor with the highest recirculation rate, showing the importance of recirculation to the degradation process. Also in these experiments anaerobic seed sludge was added. It should be noted that the leachate recirculation rates used in this work were far below the optimum rate of 310 l/ton.day (129 l/m^2.day) applied by Sponza and Ağdağ (2004) which seems to explain the consequent increase of degradation rate with recirculation rate. Here, a breaking point where higher recirculation led to flawed methanogenesis was not reached.

Hao et al. (2008) experimented with waste degradation under the condition of moisture oversaturation. This meant that the void volume of the waste was to a high degree filled with water. This is probably different from the condition in most of the reactors run by Sponza and Ağdağ (2004) and Jiang et al. (2007). They compared the regimes in two 150 cm high lab-scale reactors filled with 39 kg synthesized MSW with 55% moisture. They added water to the fresh waste to reach an oversaturated condition. The first reactor (single pass) was kept at this condition by replacing outflowing leachate with fresh water. There was no leachate recirculation here. In the second reactor leachate was recirculated at rates intermittently varying between 0 and 33 l/ton.day (0 and 14.9 l/m^2.day). The experiments were carried out at ambient temperatures varying between 10 and 22 ^oC. The single pass reactor showed with respect to leachate COD and pH the typical behavior of a conventional sanitary landfill: initially a decreasing pH (acidification) and an increasing COD, then, after about 150 days an increasing pH to about 6.8 and a strongly decreasing COD. The lowest pH value reached was about 5.2. In the recirculated reactor the leachate COD rose to about 30,000 mg/l after 3 months, but different from the single pass reactor which did not come down much. The pH remained below 6.0. This was a clear case of persistent acidification. The biogas production in the single pass reactor slowly increased with time (and temperature) and in the recirculated reactor it remained generally much lower even at the better temperature conditions later in the experimental period. Apparently, under the oversaturated conditions the recirculation had a negative impact on methanogenesis. This is a completely different situation than in the above mentioned work of Jiang et al (2007). The composition of the biogas in the two reactors was about the same. Despite the lower biodegradation and reduced gas production in the recirculated reactor, the waste settlement was bigger.

Hao et al. (2008) present an interesting theory of waste settlement. The settlement is explained through the fact that leachate release and leachate dosage at the top were not simultaneous. Release increases the void ratio and weakens the structural strength with waste settlement as a consequence. The addition of leachate at the top enhances the pressure on the waste which also pushes down the waste level. Both effects were stronger in the recirculated reactor than in the single pass reactor. Another consequence of waste settling is an accumulation of gas in the void spaces and an upwards push of liquid to the top of the reactor (which was actually observed in the recirculated reactor). The top liquid could further push down the waste. In the single pass reactor the accumulated biogas could lead to an expansion of the waste mass and therefore no waste settlement. Authors suspect that waste settlement (reduced void spaces) leads to biogas accumulation and solid-liquid separation, which reduces the biodegradation process due to hindered contact between the solids and the microorganisms. It was clear that the oversaturated condition plus the leachate recirculation caused a situation of permanent acidification and low gas production. Such a situation probably also occurred in the experiment of Sponza and Ağdağ (2004) at the highest recirculation rate of 880 l/ton.day (300 l/m^2.day), which could have caused the persistently acidified state of that reactor. This acidified state is called ensiling or souring effect.

So far the results of three publications with wet waste. A different situation occurs at landfills where relatively dry waste is deposited. Several authors reported about the impact of recirculation in this case (Chugh et al. 1998; Benson et al. 2007; Bareither et al. 2010). In the study by Benson et al. (2007) and co-workers of five North American landfills the applied recirculation rates were very low (in the order of 10 – 20 ml leachate/ton.day (21-163 l/m^2.year) and moisture content seemed to remain below field capacity. In only one case with highest recirculation rate a positive effect of recirculation on biodegradation could be ascertained. Authors presuppose that the applied recirculation rates were too low to observe impacts on biodegradation.

It can be concluded that moisture and leachate recirculation may have a positive effect on biodegradation in a landfill bioreactor (Benson et al. 2007; Jiang et al. 2007), but that biomethanation is unable to keep pace with acidification at high recirculation rates Sponza and Ağdağ (2004) and oversaturation (Hao et al. 2008). The literature does not yield a complete picture of the influence of recirculation on the mechanism of biodegradation yet. It is evident that a too low moisture content has a negative effect on all microbial activity. The addition of moisture (leachate recirculation) may lead to better distribution of microorganisms, substrates and nutrients through the waste mass, which has a positive effect on biodegradation. Other positive effects may be expected from the addition of alkalinity and methanogenic seed sludge. Chugh et al. (1998) mention the positive seeding effect of recirculation of leachate from well stabilized LFB cells through cells with fresh waste. A high moisture content, and even oversaturation, by itself does not have to lead to persistent acidification of the waste mass as the work of Hao et al. (2008) illustrates. However, the combination of oversaturation and recirculation of leachate highly loaded with volatile fatty acids causes souring accompanied by a strong reduction, though not a complete suppression, of the methane production (Renou et al. 2008; Hao et al. 2008). This phenomenon is very interesting if the aim is to deliver acidified leachate to an anaerobic leachate treatment reactor for the production of biogas. With regard to waste settlement both biological and physical factors play a role. It is evident that waste degradation and reduction of the waste mass leads to settlement. Release of leachate from the bottom of landfills and deposition on the top may have the same effect. The building up of liquid and gas in the void spaces of the waste could hinder waste settlement.

Of course, the question of design leachate recirculation rates rises. It is evident that the design rate depends on the aim: either full biomethanation or acidification. If the aim is full biomethanation within the landfill, an optimum rate probably depends on composition, particle sizes, density and porosity of the wastes and may further be influenced by the height of the landfill, the ambient temperature, addition of seed sludge and alkalinity and the impact of rainfall. In order to avoid souring oversaturation with leachate should definitely be avoided. Favorable conditions for acidification with only minor biomethanation are a high moisture content and recirculation rate (Sponza and Ağdağ 2004; Hao et al. 2008). On the basis of the articles reviewed above it can be concluded that leachate recirculation can be applied for stimulating either a more or less complete biodegradation trajectory of the waste in the landfill (including the production of biogas) or a partial biodegradation trajectory to acidified leachate which can be treated outside the landfill. No straightforward conclusions can be drawn about optimum or even a recommendable design rates of recirculation. The recirculation rates that have led to enhanced degradation vary widely: from a few to hundreds of liters/ton wet waste.day, or from little to more than 100 l/m^2.day. As all the mentioned studies (Table 4.1) refer to lab- or pilot-scale installations, one should be careful with applying their conclusions to full-scale recirculated landfills. In practice, field experimentation is recommended to determine the site specific range of recirculation rates. In landfills a positive effect might be expected from the seeding of fresh waste with microbial flora from stabilized waste.

Table 4- 1: Effect of recirculation on biodegradation of wet MSW

Type of study	Recirculation rate (l/ton.day)	Recirculation rate (l/m^2.day)	Effect on biodegradation	Source
Pilot scale Under saturated	0; 1; 4.1; 8.2	0; 4; 15; 30	Methane production increases with recirculation rate	Jiang et al. (2007)
Lab-scale Under saturated	0	0	4.4% of waste COD converted to methane[2] after 220 days	Sponza and Ağdağ (2004)
	310	129	16% of waste COD converted to methane after 220 days	
	875	300	6.4% of waste COD converted to methane (220 days), persistent acidification	
Lab-scale Over saturated	0	0	Acceleration of methanogenesis after about 150 days	Hao et al. (2008)
	33	14.9	Persistent acidification	

4.4 Leachate characteristics

4.4.1 Sanitary landfills

Factors affecting the characteristics of leachate include waste type and composition, leachate age, precipitation and seasonal weather variation. In particular, the composition of LFB leachates varies greatly depending on the age of the landfill similarly to sanitary landfills (Silva et al. 2004). Accordingly, the distinction between young, medium and old leachate is made.

The quality of leachate at sanitary landfills has been widely studied (Kjeldsen et al. 2002b) and several LFB studies have additionally investigated the effects of leachate recirculation on leachate quality (Reinhart and Basel A. 1996; Morris et al. 2003; Sormunen et al. 2008). According to Kjeldsen et al. (2002b) pollutants in MSW landfill leachate can be divided into four groups:

- Dissolved organic matter, quantified as Biological Oxygen Demand (BOD), Chemical Oxygen Demand (COD) or Total Organic Carbon (TOC), Volatile Fatty Acids (VFA) that accumulate during the acid phase of the waste stabilization and more refractory compounds such as fulvic-like and humic-like compounds;
- Inorganic macro-components: calcium (Ca^{2+}), magnesium (Mg^{2+}), sodium (Na^{+}), potassium (K^{+}), ammonium (NH_4^{+}), iron (Fe^{2+}), manganese (Mn^{2+}), chloride (Cl^{-}), sulfate (SO_4^{2-}) and hydrogen carbonate (HCO_3^{-}), phosphate ($HPO_4^{2-}/H_2PO_4^{-}$);
- Heavy metals: cadmium (Cd^{2+}), chromium (Cr^{3+}), copper (Cu^{2+}), lead (Pb^{2+}), nickel (Ni^{2+}) and zinc (Zn^{2+});
- Xenobiotic organic compounds (XOCs) originating from household or industrial chemicals present in relatively low concentrations (usually less than 1 mg/l of individual

[2] The indicated values were measured after 50 days of waste conversion.

compounds). These compounds include among others a variety of aromatic hydrocarbons, phenols, chlorinated aliphatics, pesticides, and plasticizers.

The characteristics of the landfill leachate are usually represented by the basic parameters COD, BOD, the ratio BOD/COD, pH, suspended solids (SS), ammonium nitrogen (NH_3-N), total Kjeldahl nitrogen (TKN) and heavy metals (Renou et al. 2008). Values presented in Table 4-2 and 4-3 resulted from a review study by Renou et al. (2008). They aggregated characteristics of leachates emanating from numerous sanitary landfills in 15 different countries of North and South America, Asia and Europe. It should be noted that the composition and moisture content of MSW landfilled in these sites could be different from MSW in African cities. Nevertheless, these data are considered a good starting point for understanding leachate characteristics.

Table 4-2: Characteristics of leachate from sanitary landfills

	Parameter						
Age	COD	BOD	BOD/COD	pH	SS	TKN	NH_3-N
Young	1870-70900	90-26800	0.05-0.7	5.6-9.1	950->5000	75-13000	10-13000
Medium	1180-9500	331-14366	0.07-0.33	6.9-9.0	480-784	1100-1670	743-5500
Old	100-10000	3-800	0.01-0.37	7.0-9.4	13-1600	5-1680	0.2-1590

Note: All values are in mg/l except for pH and BOD/COD ratio

Table 4-3: Metals concentration in sanitary landfill leachate

	Parameter					
Age	Fe	Mn	Ba	Cu	Al	Si
Young	2.7	0.04	-	-	-	3.72-10.48
Medium	1.28-76	0.028-16.4	0.006-0.164	0.12-0.78	-	-
Old	4.1-26	0.13-15.5	0.15	0.005-0.08	2	<5

Note: All values are in mg/l

The characteristics of sanitary landfill leachate varies between 100 mg COD/l and 3 mg BOD/l from a more than 10-year old landfill in France to about 70,000 mg COD/l and 27,000 mg BOD/l from a young landfill in Greece. It is important to note that the majority of TKN is ammonia, which ranges from 0.2 to 13,000 mg N/l. Landfill leachate is characterized by a ratio of BOD/COD of 0.70 for young leachate and a sharp decrease to 0.04 with the aging of the landfills. This is due to the release and formation of large recalcitrant organic molecules during the conversion of the solid wastes (Renou et al. 2008).

4.4.2 Landfill bioreactor

Relevant literature regarding leachate produced in landfill bioreactor with intensive leachate recirculation and possible treatment techniques for this leachate is given by (Spagni and Marsili-Libelli 2009; Li et al. 2009; Wang et al. 2010; Zhao et al. 2010). The leachate is in general characterized by a high concentration of organic matter (COD = 1,000 - 50,000 mg/l), and a relatively low BOD/COD ratio (< 0.2). Part of the COD is present as suspended and colloidal particles and part of the COD is dissolved. The concentration of COD and BOD will vary with time. It can be expected that the ratio of BOD/COD will decrease with time, this means in general with the age of the landfill bioreactor. Fresh leachate has in general a relatively high BOD/COD ratio. The leachate from a landfill bioreactor contains also a relatively high concentration of ammonia. This concentration varies in general between 500 - 4,000 mg/l. The concentration of ammonia in the leachate will not decreased due to the recirculation process of leachate. Besides the leachate contains heavy metals, halogenated organics and inorganic salts. The key problem components of the leachate are the dissolved non-biodegradable COD and the ammonia. Removal of these components is in general complicated and expensive. The quality and quantity of the excess leachate from landfill bioreactors is of much importance to the design of treatment systems and their operating costs.

4.5 Technical aspects of leachate recirculation

In this section the technical aspects of leachate recirculation are discussed. Leachate recirculation consists of injecting leachate into the waste matrix and collecting it using a leachate collection system located above the lining at the bottom of the landfill. The way leachate is injected determines the spreading over and flow through the waste mass and can therefore be crucial to the impact of recirculation. Three components are needed to add liquids to a landfill unit:

(i) a liquids storage unit;
(ii) a conveyance mechanism to deliver liquids from the storage unit to the landfill unit;
(iii) a scheme to apply liquids to the landfilled waste mass (the liquids addition system) (Townsend et al. 2008).

Storage systems include ponds and tanks, located outside the lined landfill area. Liquids can be delivered from the storage system to the landfill in a variety of fashions. Liquids can be hauled to the landfill in a tanker truck and discharged directly to the working face or to an impoundment area (Townsend et al. 2008). Liquids can also be delivered through a piping network. Once transported to the landfill, there are several techniques to apply liquids to the landfilled waste (Townsend et al. 2008).

A typical recirculation approach for relatively dry waste is to add enough liquid to bring the landfilled waste to field capacity. In order to maximize the area impacted, leachate recirculation operations should be rotated from one area to another, pumping at relatively intense rate for a short period of time, then moving to another area.

4.5.1 Leachate recirculation techniques

There are various techniques of liquid addition categorized as: (i) surface systems - tanker truck application; spray irrigation; drip irrigation; infiltration ponds; leach field; and surface trench; (ii) subsurface systems - vertical injection wells; horizontal trenches; buried infiltration galleries; permeable blankets and a combination of horizontal lines and vertical wells (Haydar and Khire 2005; Khire and Haydar 2007; Townsend et al. 2008).

(i) Surface systems

Surface addition techniques include those where the liquid is applied to the landfilled waste by application to the landfill surface. Surface application systems depend on the ability of liquids to migrate from the surface of the landfill to the underlying waste under gravity and suction forces by the waste.

Tanker trucks are often used to directly wet the working face of the landfill. The leachate is discharged by a hose or spray nozzle. Direct application of the leachate provides for good liquids distribution in the areas where it is applied. Major concerns that must be addressed with this method is prevention of off-site leachate runoff and contamination of storm water.

Using spray irrigation, leachate can be applied to the waste as it is being disposed of or to areas where the landfilled waste is already covered. The objective of spray application is often to reduce leachate volume through evaporation. Thus it may not be useful if the primary objective is to increase the waste moisture content. Spraying is accomplished using portable spray heads that can be moved around the landfill as the working face progresses or with a more permanent system for areas that will remain inactive for some time.

In drip application, drip hoses or pipes are placed at the surface of the landfill, either directly on the waste before cover placement, or in a more permanent leach bead, such as gravel. Drip application, if well designed, can provide for relatively uniform liquids distribution at the surface of the landfill.

Infiltration ponds are a shallow depths excavated into the waste to form pond walls. Ponds can be an effective method for wetting the area under the pond, but the exposed leachate poses problems with respect to control of storm water, gas control, and exposed waste. Infiltration trenches are trenches excavated at the surface of the landfill and used to distribute the liquids into the upper layers of the waste mass. The trenches are typically fitted with a liquids distribution pipe and backfilled with a porous media.

Common concerns of surface application systems are generally limited to smaller leachate application rates compared to subsurface systems. Other concerns common to surface systems include formation of aerosols and their exposure to workers, off-site migration of contaminants with storm water runoff, and gas emissions and escape to the environment from the liquids application area. For most bioreactors, surface application system will be one tool used to add liquids in addition to subsurface application approaches (Townsend et al. 2008).

(ii) Subsurface systems

In sub-surface systems, the liquids are added to the landfill using a system located within the waste. Liquids can be added under pressure maintained by a pump or by a standing head of liquid. The addition of leachate under pressure promotes distribution of liquids to areas otherwise inaccessible. Subsurface systems may be grouped as vertical and horizontal systems. These can also include a combination of techniques (vertical wells connected to horizontal layers). These devices are installed either while waste is being placed in the landfill (deep horizontal trenches) or after the waste has reached its final grade (shallow horizontal trenches, vertical wells) (Townsend et al. 2008). Horizontal trenches are more commonly used in modern lined landfills whereas vertical wells are relatively common in retrofit landfills where it is not cost effective or possible to install horizontal trenches (Haydar and Khire 2005).

Vertical wells is a type of liquids addition subsurface system whereby a number of vertical wells are constructed across the depth of the waste matrix by drilling. Previous reported applications have used well diameters ranging from 25 to 300 mm. Large diameter wells 300 to 900 mm diameter usually consist of 300 to 600 mm diameter pipe surrounded by a permeable medium such as stone or gravel (Townsend et al. 2008). Larger wells are most often installed as single wells at depths nearly the entire length of the landfilled waste. PVC pipes are commonly used for vertical wells. An advantage to PVC pipes is their ease of installation and maintenance. A major maintenance issue with vertical wells is frequent adjustment of well heights above the surface of landfill. Experience has found that differential settlement of the landfill surface with time resulted in extension of wells above the landfill surface, and that wells should be shortened periodically.

In horizontal trenches liquid addition system, a large number of horizontal trenches are placed at various levels within the landfill while the waste is placed. Horizontal pipes are embedded in permeable material within the trench. Liquid addition in these lines is generally started after at least one lift of waste has been placed over these trenches. This system is the most common practice as-built bioreactors. HDPE is the most commonly used pipe material due to its strength and durability. Typical perforation size for leachate injection lines is 6.25 to 12.5 mm with multiple holes every few centimeters. The best high permeability filler material used for bedding and cover media are gravel or rock. For cost considerations, other filler materials like mulch, tire chips or glass may also be used (Townsend et al. 2008). The advantage of the drainage media is that if the pipe ever breaks, the drainage media can still act as a conduit for liquids movement.

Infiltration blankets of a pipe embedded in a highly permeable media (or material) laid over a much larger area of landfilled waste than a buried trench. Infiltration blankets act as installed layers, typically one to two feet thick of highly permeable materials. They can be used to provide uniform distribution of recirculated leachate over the widest possible area (Townsend et al. 2008). This system is constructed as landfilling progresses and can be regarded as buried infiltration pond. Horizontal galleries or granular beds can be installed within lifts of waste to provide large areas for liquids distribution.

4.5.2 Design

The design of a landfill recirculation system depends on the aim of the landfill: full or partial biodegradation (acidification). Dependent design parameters are the initial waste moisture content, the mass of liquids to be added initially and the rate of recirculation. The type of leachate recirculation system utilized and the method of operation are selected after appropriate consideration of intended goals related to moisture distribution, minimizing environmental impact, and regulatory compliance (Reinhart et al. 2002).

Currently, there are no specific design guidelines available for designing a subsurface leachate recirculation system consisting of vertical wells (Khire and Mukherjee 2007) but Al-Yousfi (1992) and Townsend (1995) in Reinhart, McCreanor, and Townsend (2002) developed equations to assist in the design for both horizontal trenches and vertical wells. McCreanor (1998) in Reinhart, McCreanor, and Townsend (2002) used the United States Geological Survey's Saturated-Unsaturated Flow and Transport model (SUTRA) to simulate the behavior of horizontal leachate recirculation trenches and vertical leachate recirculation wells.

Horizontal trenches are constructed by excavating the surface of landfilled compacted solid waste, placing a perforated pipe in the trench, and backfilling with a permeable material. The trench is then covered, preferably with additional compacted solid waste. Al-Yousfi (1992) in Reinhart, McCreanor, and Townsend (2002) developed an equation that can be used to estimate the required horizontal distance between trenches. Townsend (1995) in Reinhart, McCreanor, and Townsend (2002) also developed equations based on uniform flow theory for saturated conditions to estimate the area influenced by a horizontal infiltration trench.

Vertical wells for leachate recirculation are constructed in the same manner as vertical wells for gas extraction, generally requiring drilling into the waste mass and installation of piping. In some cases, wells are constructed as waste is placed, by installing pipe sections at each waste lift. Al-Yousfi (1992) in Reinhart et al. (2002) proposed that the radius of influence of a well, defined as the maximum distance of leachate movement from the well, could be estimated based on a mass balance of the leachate. It was then concluded that wells should be spaced no more than 60 m apart to ensure efficient wetting of the waste mass. Vertical wells should be spaced at approximately twice the indicated lateral movement and distanced at least the indicated lateral movement from the landfill boundaries.

4.6 Leachate treatment

4.6.1 Overview of options

Leachates from conventional landfills and landfill bioreactors show strong variation in amount and concentration of the pollutants. These variations are due to the variations in the technical modification of conventional landfills and landfill bioreactors, mode of operations of the these landfills and landfill bioreactors, type of waste, pre-treatment of the waste before disposal into the landfill or landfill bioreactor, climate (temperature and precipitation), age of the landfill and leachate, leachate recirculation factor, pre-treatment of the leachate before recirculation, etc. All these aspects influence the amount and the composition of the leachate.

In general it can be stated that the amount and composition of the leachate is strongly determined by the specific and unique character of each landfill or landfill bioreactor. It means that the treatment process that is required is also very specific for a certain landfill. Besides, the required treatment process also strongly depends on the set standards for discharge of the treated leachate into a sewer system or onto surface water. Here, we will focus primarily on the treatment of leachate from a landfill bioreactor (with an intensive recirculation of the leachate). For each landfill a tailor-made treatment technology has to be designed. The biggest challenges in landfill leachate treatment are removal of high concentrations of non-biodegradable (refractory) organic compounds and ammonia. Landfill leachate treatment technologies for sanitary landfills and landfill bioreactors can be classified into four major groups (Renou et al. 2008; Abbas et al. 2009):

(a) combined treatment of leachate and domestic sewage in municipal wastewater treatment plants;
(b) biodegradation by means of aerobic and anaerobic processes;
(c) chemical and physical methods: chemical oxidation, adsorption, chemical precipitation, coagulation/flocculation, sedimentation/flotation and air stripping;
(d) membrane filtration: microfiltration, ultrafiltration, nanofiltration and reverse osmosis.

It should be noted that most of the next discussion is based on literature that deals with sanitary landfills or landfill bioreactors from the U.S.A and Europe.

4.6.1.1 Combined treatment of leachate and municipal wastewater

This treatment technique, sometimes called channeling, involves landfill recycling of leachate at the site and treatment of the surplus leachate at a municipal wastewater treatment plant or, alternatively, disposal via a sewer outfall into the sea. This practice was preferred for its easy maintenance and low operating costs (Lema et al. 1988; Ahn et al. 2002). However, this option has been increasingly questioned (Cecen and Aktas 2004) due to the presence of organic inhibitory compounds with low biodegradability and heavy metals (typical characteristic of leachate from LFBs) that may reduce treatment efficiency. An argument in favor could be that neither nitrogen (from leachate) and phosphorus (from sewage) need to be supplied at the treatment plant (Abbas et al. 2009). If there is sufficient capacity for removal of nitrogen and phosphorus at the municipal wastewater treatment plant, removing them together in the treatment system is plausible. In a few studies authors have tried to optimize the volumetric ratio of leachate in the total wastewater flow. Combined treatment was investigated by Diamadopoulos et al. (1997) using a Sequencing Batch Reactor (SBR) operated with filling, anoxic and settling phases. De Velasquez and co-workers (2011) found that landfill leachate could be successfully co-treated in facultative lagoons (Mexico) through admixture of 10% of leachate to 90% low-strength but saline municipal sewage. The average COD and BOD_5 values of the leachate were about 5,800 and 875 mg/l respectively. The design hydraulic retention time of the system was 17 days. The obtained BOD_5 removal varied from 70 – 78%. Moreover, the effluent quality may need to be improved with Powdered Activated Carbon (PAC), particularly if the volumetric leachate input exceeds 10% of the total flow (Cecen and Aktas 2001; Abbas et al. 2009).

4.6.1.2 Evaporation

In tropical climates evaporation could help in reducing the quantities of wastewater to be treated. Evaporation is a process that accompanies and reinforces the treatment in stabilization ponds and constructed wetlands. The evaporation rate from an open water surface depends on temperature and wind conditions and is expected to be in the range of 5 to 10 mm/d. or 50 – 100 m^3/ha.day. Due to rainfall there will be a net addition of water of about 800 - 1000 mm/year (typical annual rainfall in Tanzania) corresponding to an evaporation period of 80 – 160 days. Evaporation ponds have to be shallow in the order of 0.5 m water depth. From time to time evaporation ponds should be emptied and the remaining highly polluted sludge be deposited in a safe way. Evaporation ponds should be well lined to protect soil and groundwater.

4.6.1.3 Biological treatment

Anaerobic biological leachate treatment techniques include several high-rate anaerobic treatment techniques, such as UASB reactors, anaerobic filters, hybrid bed filters and fluidized bed reactors. Aerobic treatment processes include, among others, lagooning systems, facultative ponds, activated sludge processes, constructed wetlands, trickling filters and moving bed biofilm reactors. Biological processes have been shown to be very effective in removing organic matter from immature leachates when the BOD/COD ratio has a high value (> 0.5). Due to its reliability, simplicity and high cost-effectiveness, biological treatment is commonly used for the removal of the bulk of the biodegradable organic pollutants in leachate from sanitary landfills (without recirculation). Nitrification and de-nitrification for ammonia-nitrogen removal require combined aerobic and anoxic biological treatment processes. De-nitrification of nitrified leachate will require addition of biodegradable matter as this leachate presumably contains too little of this matter. From a process operation point of view post-denitrification is probably more simple than pre-denitrification. With increasing age of the landfill site, the major presence of refractory compounds in the leachate (mainly humic and fulvic acids) tends to limit the effectiveness of biological treatment processes (Abbas et al. 2009). In landfill leachate treatment a common combination of treatment technologies are UASB pre-treatment followed by activated sludge secondary treatment with nitrification and de-nitrification and post-treatment in facultative and aerobic stabilization ponds.

Facultative and aerobic ponds (lagoons) derive their strengths from the photosynthetic oxygenation by algae during daylight and the symbiosis between algae and bacteria. These ponds are usually 1 to 2 m deep. The permissible organic loading rates of facultative and aerobic ponds increase with the ambient temperature. This makes this treatment technology most suitable for tropical climates. For municipal waste water the maximum BOD loading rate of facultative ponds is about 400 kg BOD/ha.day at 25^o C (Arthur 1983). Facultative ponds are capable of nitrogen removal through a combination of processes and phosphorus removal. Nitrogen removal is brought about by volatilization of ammonia, nitrification and de-nitrification and uptake in algae and bacteria. Frascari and co-workers (2004) found that a series of anaerobic and facultative lagoons with a total retention time of 240 days (Italy) are able to remove a significant part of the main pollutants from landfill leachate. Ten-years' average COD, BOD and TKN effluent concentrations after lagoon treatment were 2,960, 470 and 370 mg/l respectively, showing overall removal percentages of 40, 60 and 77%. It is evident that the required long retention times in the treatment of leachate lead to large pond areas.

The effluent of stabilization ponds usually contains suspended solids mainly algal matter. The TSS concentration may exceed 100 mg/l (Reed et al. 1995, p.119). This suspended matter can be removed by maintaining a duckweed covered zone in the pond provoking a shading-out effect. Alternatively, the algae can be removed by micro-screening and soil filtration, e.g. in vertical flow constructed wetlands.

Surface and sub-surface flow constructed wetlands are used for (post-) treatment of landfill leachate. Based on literature review Vymazal and Kröpfelova (2009) showed that subsurface-horizontal flow constructed wetlands applied to landfill leachates with COD and BOD influent concentrations in the range of 1,000 and 200 mg/l respectively produced removal percentages of 25% (COD) and 33% (BOD). The average of the hydraulic loading rates of these wetlands was 2.7 cm/d and the organic surface loading rates (kg BOD/ha.day) were in the range of 50 – 100 kg/ha.day (5 – 10 g BOD/m^2.day).

A good nitrification in horizontal-flow constructed wetlands is possible. The maximum load to achieve nitrification should not exceed 0.2 g TKN/m^2.day (Platzer 1998). This means that in treatment of leachates the necessary bed area is extremely large.

In a comparison of subsurface horizontal and vertical- flow constructed wetlands at lab-scale Yalcuk and Ugurlu (2009) worked with leachate COD and NH_4-N concentrations of respectively about 260 and 120 mg/l. In vertical-flow wetlands they found COD and NH_4-N removals of 10 - 40 and 60 - 70% respectively. The hydraulic surface loading rate amounted to 2.0 cm/day and the N-loading rate 2.4 g/m^2.day. The NH_4-N removal percentages of horizontal flow wetlands were in the range of 30 – 50% and thus were lower than in vertical-flow wetlands. It may be noted that subsurface-flow constructed wetlands show a combination of treatment processes including screening, filtration, aerobic, anoxic and anaerobic biological conversion, precipitation, adsorption and uptake in the macrophytes growing on the filter bed. With regard to landfill leachate treatment subsurface vertical- flow wetlands are in particular effective in removal of suspended and colloidal solids, conversion of ammonia to nitrate and de-nitrification and removal of phosphate. The removal of recalcitrant COD compounds is limited. As permissible hydraulic and organic loading rates are low the spatial requirements of constructed wetland are high. For the treatment of 100 m^3/d at a hydraulic loading rate of 2 cm/day (20 l/m^2.day) a surface area of 0.5 ha is needed.

4.6.1.4 Physical/chemical treatment

Physical and chemical processes include physical and chemical methods for the removal of suspended solids, colloidal particles, floating material, color, ammonia-nitrogen and toxic compounds. These methods imply processes such as flotation, coagulation/flocculation, precipitation, adsorption, absorption, chemical oxidation and air stripping (Renou et al. 2008). Physico-chemical methods are used along with biological methods mainly to improve the total treatment efficiency when the biological oxidation process is hampered by the presence of bio-refractory materials (Abbas et al. 2009; Wiszniowski et al. 2006). An often mentioned physico-chemical treatment option for leachate is lime coagulation and flocculation (colloidal particles removal) combined with air stripping for removal of ammonia followed by granulated activated carbon (GAC) filtration to remove refractory organics and heavy metals.

4.6.1.5 Membrane filtration

Membrane filtration is a specific physical treatment technique. Membrane filtration techniques include microfiltration, ultrafiltration, nanofiltration and reverse osmosis. Microfiltration is a low pressure cross-flow membrane process for separating colloidal and suspended particles. Microfiltration can be used as stand-alone technology but is often used as a pre-treatment process for other membrane processes such as nanofiltration or reverse osmosis or in combination with chemical treatment processes (Abbas et al. 2009). In recent years membrane treatment, though often considered expensive, has become more and more accepted as it can deliver high quality effluents that other methods hardly achieve.

4.6.2 Guidance to the choice of leachate treatment options

In the selection of a leachate treatment technology, the crucial issue is that the chosen technology be appropriate to the situation under study. An appropriate technology is selected by matching the characteristics of the situation with those of the technology. The most relevant characteristics of the situation are:

- Average ambient temperature
- Quantity of leachate
- Leachate strength (COD, BOD, NH_3-N, heavy metals, minerals, toxic organics, etc.)
- Leachate age (important for the fraction of refractory compounds)
- Discharge standards the treatment process has to satisfy
- Availability of space
- Available arrangements for operation and maintenance
- Available funds for investment and operation and maintenance

The *ambient temperature* determines the feasibility of some biological treatment techniques. A very low temperature excludes methods like stabilization ponds and anaerobic treatment. High ambient temperatures favour the application of anaerobic treatment. High quantities of leachate make combined treatment of leachate in a municipal sewage treatment plant and the application of expensive biological, physical and chemical methods less appropriate.

For *high strength leachate* in combination with a sufficiently high BOD/COD ratio anaerobic pre-treatment is preferable. At lower concentrations a wider spectrum of biological and physico-chemical methods is suitable.

Old leachate is characterized by a high concentration of refractory compounds and a low BOD/COD ratio. Here, combined treatment in municipal wastewater treatment plants and physico-chemical treatment, e.g. with membranes preceded by adequate removal of colloidal particles, are the preferred methods.

For *young leachate* with its high concentrations of biodegradable substances biological treatment methods are preferred. If however, discharge standards are strict, the biological treatment should be followed by a relevant physico-chemical post-treatment process. Table 4-4 offers a summary of the effectiveness of the various types of treatment measured against the age of leachate from sanitary landfills in which the leachate is not recirculated.

Stringent *discharge standards* usually favour the application of treatment trains consisting of multiple treatment stages. Relatively lenient discharge standards permit the application of biological treatment only, like stabilization ponds or anaerobic plus aerobic treatment.

A lack of *space* precludes the application of stabilization ponds and constructed wetlands which have a low surface loading rate.

The *operation and maintenance* of leachate treatment technologies comes with certain institutional requirements. It is assumed here, that physico-chemical methods require more operational skills and swift provision of spare parts and chemicals than the other methods. Also aerobic and anaerobic treatment will not function durably without very regular maintenance of plants. Combined treatment, recirculation, lagooning and constructed wetlands are judged relatively simple methods that do not pose high requirements to the operational organization. The question of operation and maintenance requirements cannot be seen in isolation from costs. In general, higher discharge requirements will lead to higher investment and operational costs of leachate treatment. Combined treatment and landfill recirculation probably will bring relatively low leachate treatment costs. The costs of lagooning and constructed wetlands depend to a high degree on the cost of land. Based on literature about costs of treatment plants, it is assumed here, that the leachate treatment methods rank in order of increasing costs as follows: Recirculation < Combined treatment < Lagooning < Anaerobic treatment < Aerobic (mechanized) treatment = Constructed wetlands < Physico-chemical treatment (van Buuren 2010, chapter 7).

Table 4-4: Effectiveness of leachate treatment versus leachate age (adapted from (Abbas et al. 2009))

Type of treatment	**Leachate age**			**Target of removal**	**Remarks**
	Young	**Medium**	**Old**		
Channeling					
Combined treatment with domestic sewage	Good	Fair	Poor	Removal SS	Excess biomass and nutrients
Recycling (Recirculation)	Good	Fair	Poor	Improve leachate quality	Least expensive and low efficiency
Biological					
Aerobic processes	Good	Fair	Poor	Removal SS	Hampered by refractory compounds and excess biomass
Anaerobic processes	Good	Fair	Poor	Removal SS	Hampered by refractory

					compounds, long time and gas
Physico/chemical					
Coagulation/flocculation	Poor	Fair	Fair	Heavy metals and SS	High sludge production and subsequent disposal
Chemical precipitation	Poor	Fair	Poor	Heavy metals and ammonia-nitrogen	Requires further disposal due to sludge generation
Adsorption	Poor	Fair	Good	Organic compounds	Carbon fouling can be a problem and GAC adsorption is costly
Oxidation	Poor	Fair	Fair	Organic compounds	Residual O_3
Stripping	Poor	Fair	Fair	Ammonia-nitrogen	Requires additional air pollution control equipment
Ion exchange	Good	Good	Good	Dissolved compounds cations/anions	Used as polishing step after biological treatments and treatment cost is high
Membrane filtration					
Microfiltration	Poor	-	-	Suspended solids	Used after metal precipitation
Ultrafiltration	Poor	-	-	High molecular weight compounds	Costly and limited applicability due to membrane fouling
Nanofiltration	Good	Good	Good	Sulphate salts and hardness ions	Costly and requires lower pressure than reverse osmosis
Reverse osmosis	Good	Good	Good	Organic and inorganic compounds	Costly and extensive pretreatment required

Table 4-5 reviews the suitability of various treatment methods for sanitary landfill leachate under different conditions. An indication with a plus (+) judges that the treatment method is favorable to the described situation; an judgment with (0) judges that the method is indifferent to the characteristic of the situation and (-) indicates that the method is unsuitable to the given situation. This table has an indicative character only as absolute figures about the situation characteristics are not given.

Table 4-5: Indicative guidance for leachate treatment choice (adapted from Lema et al. (1988)

Situation characteristics				**Biological**			**Physico-chemical**
		Combined treatment	Evaporation	Lagooning	Aerobic High-rate	Anaerobic High-rate	
Ambient temperature	H	0	+	+	+	+	0
	M	0	-	+	0	+	0
	L	0	-	-	-	-	0
Quantity of leachate	H	-	-	0	0	-	-
	M	-	-	0	0	0	0
	L	+	+	+	0	0	0
Leachate strength	H	-	+	0	-	+	-
	M	0	+	+	0	+	-
	L	+	+	+	+	0	+
Leachate age	O	+	+	-	-	-	+

	Y	-	+	+	+	+	0
Discharge standards	S	-	-	-	-	-	+
	M	-	-	-	-	-	+
	L	+	+	+	+	+	-
Space availability	H	0	+	+	0	0	0
	M	0	-	0	0	+	+
	L	+	-	-	+	+	+
Available arrangements for O and M	H	0	0	0	+	+	+
	M	0	0	0	+	-	-
	L	-	+	+	-	-	-
Availability of funding	H	0	0	0	0	0	+
	M	+	+	+	+	+	+
	L	+	+	+	+	+	-

H: high; M: medium; L: low; S = stringent discharge requirements; L = lenient discharge requirements; O: old leachate; Y: young leachate.

+ : favorable; 0: indifferent; - : unfavorable.

The selection of feasible leachate treatment methods for East Africa is discussed in the next sub-section.

4.6.3 LFB leachate treatment technologies for East Africa

The selection of appropriate treatment options for LFB leachate treatment in East Africa would require a matching of the situation characteristics specified in sub-section 4.6.2 with the properties of the various treatment technologies. Tentatively, we may conclude that in East Africa the climate conditions are tropical, the quantities of leachate will be considerable, leachate strength from LFBs is expected to be high or medium and that the leachate will consist of a mixture of young and old leachate. Removal of persistent COD and ammonia are the most challenging processes (Ahn et al. 2002; Alvarez-Vazquez et al. 2004). Not much is known about the discharge requirements put to the effluent of leachate treatment plants. Here, it is assumed that the requirements will be set in agreement with what is achievable with best treatment technologies that do not involve excessive costs. If that would not be the case, advanced, relatively expensive and maintenance intensive processes are needed for the removal of persistent COD and nitrogen compounds. Such technologies are membrane filtration and advanced chemical oxidation. These are deemed less appropriate for application in East Africa. Space is not considered as a limiting factor. The reliability of operation and maintenance services is estimated to be low. The availability of funds for investment and operation are assumed to be low as well.

Applying Table 4-4 one may conclude that combined treatment is a less appropriate option due to the high quantities of leachate caused by the high initial moisture content of the waste and the scarcity of municipal sewage treatment plants. Physico-chemical treatment methods and especially membranes would probably be too expensive and require a level of maintenance that is difficult to provide in a durable way. High-rate aerobic treatment, evaporation, lagooning (stabilization ponds) and sub-surface flow constructed wetlands could be judged suitable for leachate treatment in East Africa at this moment and for the next five to ten years. However, the costs of constructed wetlands could be prohibitively high if the gravel and graded sand needed as filling material are not available on-site.

4.6.4 Leachate treatment options for East Africa

In this sub-section the treatment technologies proposed in the previous subsection are combined into three possible system options. Besides these options we will propose also a fourth more advanced high-rate technology (excluding membrane treatment). For the design of these options the expected concentration ranges of COD, BOD and NH_4N in leachate have to be assessed.

4.6.4.1 Leachate treatment Option 1: Activated sludge process and constructed wetland

This option (Figure 4-1) is comprised of the following units and processes:

- Activated sludge (AS) process
- Vertical-flow constructed wetland system (VF-CWS)

The Activated sludge (AS) process is aimed at oxidizing the remaining organics (biodegradable COD) and ammonia in the leachate. The leachate is discharged into pramiry clarifier then to an aeration basin for oxidation of the remaining biodegradable organics and particularly the nitrification of ammonia-nitrogen. After the AS process, the partially treated leachate is partly recirculated to the landfill bioreactor for denitrification of the formed nitrate. A VF-CWS is added for effluent polishing.

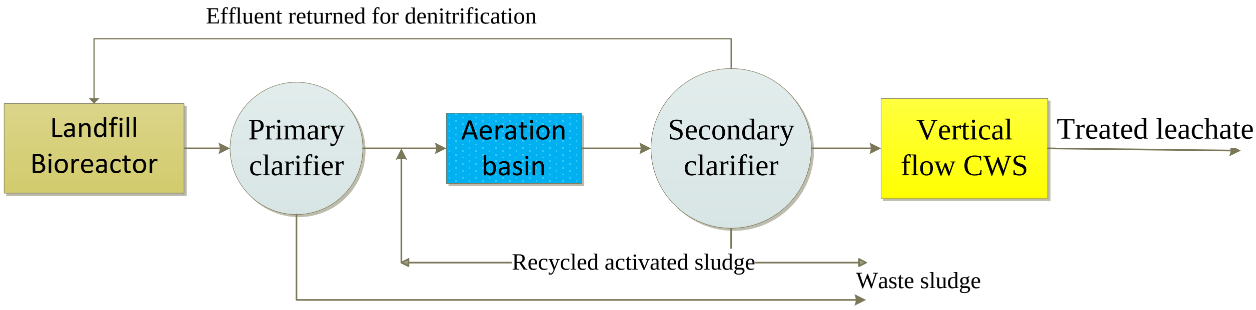


Figure 4-1: Leachate treatment Option 1: Activated sludge process and vertical flow constructed wetland system.

Strengths of the option AS plus VF-CWS

- Allows good nitrogen removal if degradable COD is low
- Capable of handling shock loads

Weaknesses of the option AS process plus VF-CWS

- High aeration cost
- High investment costs if bedding material of VF-CWS is not locally available
- Risks of biomass instabilities like sludge bulking
- Slight removal of non-biodegradable soluble COD
- Large footprint of the VF-CWS if N removal in previous stages is low

4.6.4.2 Leachate treatment Option 2: Pond system coupled with vertical flow constructed wetland system

This option involves the use of facultative ponds and then a vertical flow constructed wetland system as shown in Figure 4-2. The ponds are used to aerobically biodegrade any organics still left in the leachate after recirculation in the LFB and the wetland system for additional nitrification of the ammonia-nitrogen remaining in the pond effluent.

.

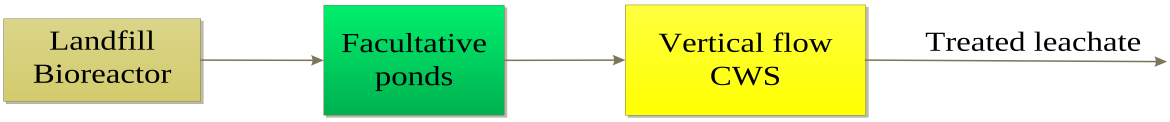


Figure 4-2: Leachate treatment plant Option 2: Pond system and vertical-flow constructed wetland system

Strengths of the option FP plus VF-CWS

- Low operation and maintenance requirements and costs
- Capable of handling shock loads

Weaknesses of the option FP plus VF-CWS

- Slight removal of non-biodegradable soluble COD
- Very large footprint
- High investment costs if bedding material of VF-CWS is not locally available

4.6.4.3 Leachate treatment Option 3: Evaporation

This option is comprised of large shallow basins. The aim is to expose the leachate to direct sun light and rely on high temperatures experienced in tropical countries. Bypassing of basins must be possible to permit full drying of waste solids. The product that remains can be used deposited in the LFB

Strengths of the option Evaporation

- Very simple process
- Very low operation and maintenance requirements and costs
- Capable of handling shock loads

Weaknesses of the option Evaporation

Very large footprint.

4.6.4.4 Leachate treatment Option 4: High-rate treatment chain for leachate of a landfill bioreactor with leachate recirculation

The described high-rate treatment system (Figure 4-3) is characterized by the application of intensive treatment steps with relatively short treatment times and the absence of facultative ponds and vertical-flow constructed wetlands. Li et al (2009) give a treatment scheme for leachate of a (conventional) landfill that exists of a combination of the following treatment steps: sequencing batch reactor serving as a primary treatment, coagulation, treatment with a Fenton oxidation system and finally treatment with an up flow biological aerated filter (UBAF).

The final concentration of COD and NH_3 obtained with this process was < 100 mg/l and < 3mg/l respectively. In case of a landfill bioreactor with recirculation of leachate it can be expected that the ratio BOD/COD is much smaller than 0.2. Compared with the previous treatment scheme some adaptation/modification is necessary some adaptation/modification. The following treatment steps can be considered.

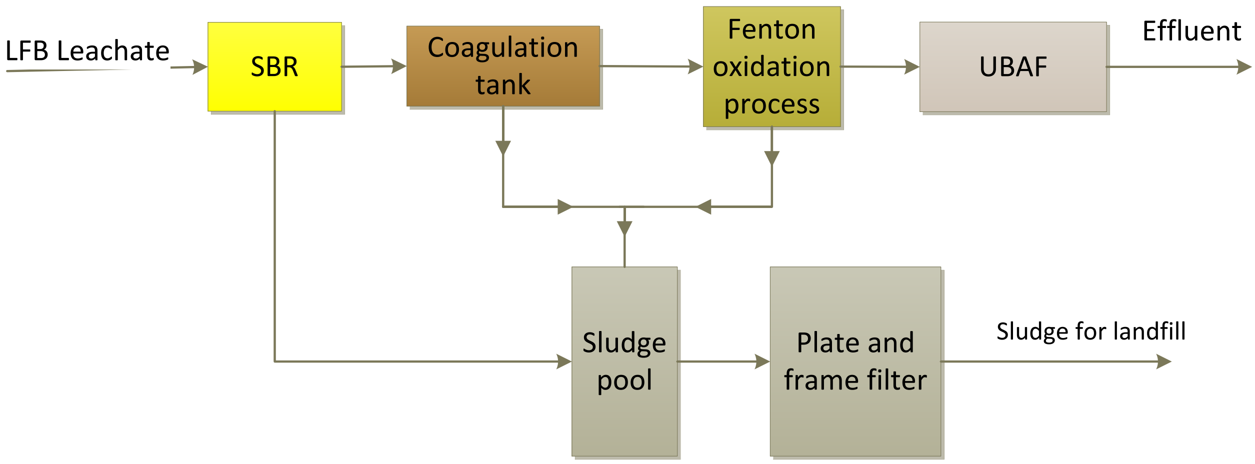


Figure 4-3: Leachate treatment Option 4: Combined biological and physico-chemical treatment

The first step is a sequencing batch reactor (SBR). With this reactor modification ammonia can be removed by conventional nitrification/denitrification. It can be expected that the concentration of biodegradable organics is too low for the denitrification step. In that case it might be possible to collect separately a small amount of fresh leachate (with a relatively high concentration of BOD) which can be obtained from the cells which have recently put in operation and to add this fresh leachate to the leachate that is treated in the sequential batch reactor.

In the SBR also the biodegradable COD that is not used in the denitrification step is removed. It can be expected that the effluent leaving the SBR has only a very low NH_3 and BOD concentration.

The second step is a coagulation/flocculation process followed by a mechanical separation process for removal of particles. This mechanical separation process can be a sedimentation tank or a flotation device. The aim of this process is primarily to remove colloidal and suspended particles, (mainly organic particles) still present in the effluent of the SBR. In this process also part of dissolved COD and BOD is removed by absorption to the particles. Because the performance of the coagulation /flocculation process also depends on the pH a pH adjustment might be necessary.

The third step is the application of a Fenton oxidation process. The aim of this process is to remove part of the dissolved COD by complete oxidation. With the Fenton oxidation process it is also possible to break down the molecular structure of non–biodegradable organic compounds in such a way that the converted molecules become more biodegradable. Important parameters that influence the performance of the Fenton oxidation reactor are pH, treatment time and concentration of the Fenton reagents (ferrous sulphate and hydrogen peroxide). Besides complete oxidation of COD and transfer of non-biodegradable organics the Fenton oxidation process results in the formation of particles which can be removed from the effluent by means of coagulation/flocculation/sedimentation.

The fourth step is an aerobic biological treatment step to remove biodegradable organics which have been produced in the Fenton oxidation step. To this aim an up flow biological aerated filter can be used. This process is followed by a separation process for the residual of colloidal and suspended particles. Based on literature references it can be expected that with this process chain a final concentration of COD < 100 mg/l can be obtained and a final NH_3 concentration <3 mg/l.

4.6.4.5 Other treatment possibilities

Alternative treatment schemes are possible. Instead of a SBR also a standard continuous flow nitrification/denitrification reactor can be applied as in option 1. To remove ammonia by means of a conventional nitrification/denitrification process a biodegradable carbon source is necessary and also a relatively large amount of electrical energy for the aeration process. In case of high ammonia concentrations the treatment costs are relatively high due to the large amount of electrical energy that is necessary and the need of a biodegradable carbon source, also representing a certain amount of electrical energy. However, it is also possible to apply more advanced energy saving and more cost-effective biological ammonia removal processes (Wang et al. 2010). These processes are based on partial nitrification of the ammonia to nitrite followed by anaerobic ammonium oxidation of the residual ammonia concentration (ANAMMOX). Several modifications of these advanced processes for removal of ammonia exists. However, the experience with treatment of leachate is limited. In general these processes are more difficult to operate and the final concentration of N-components that can be obtained is relatively high. The processes are especially of interest in treating wastewaters with relatively high ammonia concentrations.

4.7 Conclusions

1. The water content of the municipal solid waste from East Africa is very high (above 60%) so that the amount of leachate released in LFB processes are expected to be considerable;

2. Leachate recirculation in the LFB stimulates microbiological conversion processes and controls the type of microbiological processes responsible for waste degradation;

3. To stimulate biogas production in the LFB under East African conditions (high moisture content) the leachate recirculation rate in LFBs has to be in the order of 10 mm/day. In this way the recirculated leachate can reach the waste at all depths in a landfill cell within an acceptable time;

4. To keep the leachate from an LFB in the acidified form the required recirculation rate of leachate is in general higher than in case of full methanogenesis. Control to keep the concentration of volatile acids below a certain level is required to keep the hydrolysis and acidification process going;

5. Key components in development and the assessment of treatment processes for leachate are: NH_3 and non-biodegradable soluble COD;

6. The composition of leachate strongly changes with the age of the landfill. Leachate from young landfills is in general characterized by a relatively high value of BOD/COD. Leachate from old landfills has a relatively low value of BOD/COD and a high value of NH_3;

7. The extent of leachate treatment strongly depends on the effluent discharge standards that have to be met and the point of discharge;

8. Taking into account the physical and societal conditions of Tanzania and East Africa the following leachate treatment options are deemed feasible:
   - Activated sludge process coupled with constructed wetlands;
   - Pond system (facultative ponds) coupled with constructed wetlands;
   - Evaporation;

9. Under stringent effluent requirements, in the future a combined biological physico-chemical treatment process could be applied. This process could consist of sequencing batch activated sludge treatment, coagulation/flocculation, chemical oxidation and aerobic biological post-treatment.

CHAPTER 5

Modeling of landfill gas

5.1 Introduction

A landfill is a very complex heterogeneous environment (Zacharof and Butler 2004) and landfill processes are almost impossible to analyze in a deterministic way (Powrie and White 2004) thus present considerable modeling challenges. Landfill models based on reasonably representative landfill bio-chemical processes can be used to carry out calculations, but the results can only to a limited extent be expected to provide an accurate simulation of the events taking place in materials as complex and heterogeneous as landfilled waste. The value of a landfill model is that it provides an effective methodology for organizing and assessing complex biochemical datasets. Modeling is useful in that it can highlight inconsistencies and provide an insight into where the focus of research attention should be (Powrie and White 2004). All modeling attempts are geared towards understanding of the following processes in landfills: 1) landfill gas generation and the economic viability of gas recovery; 2) the rate at which the waste mass degrades or otherwise changes towards a stable, non-polluting state in equilibrium with the environment; 3) the pollution potential remaining within the landfill at any given time; 4) rates and amounts of waste settlement as basis for the design of engineered containment and control features; 5) the hydro-geological properties of the waste, which influence the way in which water flows through the landfill and interacts with the environment (Powrie and White 2004).

The focus of this chapter is on modeling of landfill gas (LFG), a mixture of methane and carbon dioxide, generated as a result of waste landfilled whereby the organic fraction in the waste decomposes. The generated LFG contributes significantly to greenhouse gas emissions. In this chapter a literature review encompassing models for quantification of methane generation from an landfill bioreactor and a critical evaluation of the models is discussed. Eventually, innovative modified models of methane generation and some theoretical calculations for one cell and many cells with waste landfilled over a specified period of time are presented. On the basis of these models the LFG generation under different scenarios are calculated in chapter 7.

5.2 Existing LFG generation models

5.2.1 General

Several methods have been described for modeling landfill gas formation. In the 1970s researchers began model development for prediction of gas recovery for both sanitary landfills and landfill bioreactors but the type of landfill considered was not always clear. The researchers and other investigators developed qualitative and quantitative descriptions of the LFG generation process based on available but limited landfill data (Oonk 2010). In the 1980s the first LFG formation models were developed to address the landfill gas recovery projects and the amount of gas that can be formed and expected for the next 10 years.

In the mid-90s modeling emphasis shifted to quantification of methane emissions, first on a national scale and later on landfill-by-landfill basis in the framework of E-PRTR (European Pollutants Release and Transfer Register). A number of emission models were then developed. In 1996 the U.S. EPA promulgated regulations (amended in 1998) calling for the control of landfill gas emissions. As part of these regulations, the U.S. EPA developed a methodology for determining landfill gas generation (USEPA 2005b). Additional models were developed in the framework of the reporting obligations for greenhouse gas emission accounting e.g. in the framework of UN-FCCC. However, different emission models give very different results with individual landfills, even when the same data is entered. In both sanitary landfills and LFBs, the organic matter in the landfilled waste is converted to LFG. The major difference between a sanitary landfill and a LFB is the enhancement of biodegradation rate of the organic matter in the LFB by recirculation of leachate, which results in rapid generation of LFG, recovery of airspace and in-situ leachate treatment. The determination of LFG generation potential and rate is crucial as these are the most important parameters for sizing the gas collection and control system, the flaring system or the electric power plant, evaluation of greenhouse gas emissions and global warming potential. Many factors influence the generation of LFG in a LFB, but the most important ones are the presence of degradable organic components in the waste, the moisture content, the age of the residue, the pH, temperature and the organic loading via the recirculated leachate (Jiang et al. 2007; Machado et al. 2009) as applied in a LFB. Organic matter is not a single homogenous component, but consists of a different constituents with degradability varying from slow to fast. Practically not all organic matter that can be converted to LFG will be converted because conditions in parts of the reactor inhibit methanogenic activity. Methane generation potential is based on the total amount of organic matter present but corrected for the amount of organic matter that does not degrade under anaerobic conditions and the amount that does not degrade due to presence of unfavorable conditions mainly brought about by reactor design, operation and climatic conditions.

In general, landfill gas formation models are not based on microbiological or biochemical principles, but more on a practical and empirical description of formation as observed in laboratory experiments or in full-scale recovery projects. Most LFG production models are based on municipal solid waste (MSW). They are therefore not automatically suitable for situations with reduced amounts of organic waste (Scharff and Jacobs 2006). Input parameters in these generation models is the amount of waste landfilled in each year of utilization and in most models also a specification of the waste. LFG generation models provide a description of the total amount of LFG formed during the lifetime of the landfill as a function of time. Such a model describes how the potential is released in the 1st, 2nd, 3rd year and so on until closure of the landfilled cells. Most models are built on first-order kinetics and some are single-phase while others are multi-phase models.

The next sub-sections give an overview of the following selected models: LandGem model (USEPA) (USEPA 2005b); First order model (TNO) (Oonk and Boom 1995; Scharff and Jacobs 2006), Multiphase (Afvalzorg) (Scharff and Jacobs 2006) and other models including GasSim (Golder Associates 2010 for the Environment Agency) (Gregory et al. 2003b; Scharff and Jacobs 2006); French E-PRTR (Oonk 2010); and EPER model France (ADEME) (Scharff and Jacobs 2006).

5.2.2 LandGEM

Currently, the LandGEM model is the most used LFG emission model. This model is similar to the U.S.EPA model. The most recent version of the model is the 3.02-version, dated May 2005. US EPA protocols (USEPA 2005a) state that the composition of waste used in this model reflects US waste composition of MSW, inert material and other non-hazardous wastes. A disadvantage of LandGEM is that it cannot allow for differences in the characteristics of the organic matter because it considers all waste to be MSW. For a landfill containing non-biodegradable waste the amount of inert material (i.e. ash) may be subtracted from the total amount of waste. However, LandGEM recommends subtracting inert materials only when documentation is provided and approved by a regulatory authority. Subtraction is not recommended for sites that are typical MSW landfills containing a range of wastes that may or may not be degradable (Scharff and Jacobs 2006).

The basis of the Land GEM model is to subdivide the disposal site in sections filled with waste each with a certain age. It is assumed that for each section the microbiological conversion of waste to methane is given by a first-order degradation of the total amount of waste following the amount of waste present at time t with a reaction rate constant k_L.

LandGEM assumes the first-order methane production rate equation to estimate annual methane production (m^3 CH_4/year) proportional to the total amount of waste. As mentioned above no distinction is made between organic biodegradables and inert wastes. For this model, the potential generation capacity of 96 m^3 methane per ton of waste in a Wet landfill (Bioreactor) and 100 m^3 methane per ton of waste in conventional landfills is used (Table 5-1). The model assumes that the produced biogas is 50% v/v methane.

The annual methane generation rate is proportional to the sum of the production rate of methane of the various sections i of different age:

$$\approx \sum \text{methane production rate of section i}$$

$$\approx \sum \mathrm{Lo} * \mathrm{Mi} * \mathrm{k_L} * \mathrm{e}^{-\mathrm{k_L} * \mathrm{t_i}}$$

Where;

L_o methane generation potential (m^3 CH_4 /ton waste);
M_i original amount of mass of waste disposed in section i (ton waste) ;
k_L LandGEM reaction rate constant, the methane generation constant ($year^{-1}$);
t_i the lifetime of section i (year).

The model is built on separate default values for methane generation rate constants for conventional regions, arid regions and for enhanced degradation cells of wet (bioreactor) landfills as shown in Table 5-1.

Table 5-1: Values for methane generation rate constants k_L and potential generation capacity

Landfill type	**k_L value** ($year^{-1}$)	**Potential generation capacity** (m^3CH_4/ton of waste)
Conventional	0.04	100
Arid Area	0.02	100
Wet (Bioreactor)	0.7	96

Source: LandGEM v.302 Guide

5.2.3 TNO model

This is also a first-order model that describes landfill gas generation as a function of total amount of waste deposited from different origins (household waste, industrial waste, etc.)

with a first-order reaction constant of the biogas production process (k_T) with respect to the biodegradable organic carbon. The parameters of the TNO-model are based on real data of landfill gas generation. The model considers one site with waste of a certain age and production of LFG as biogas. The effect of age is accounted for in the first-order decay model. As for the LandGEM model, it is assumed that the degradation rate of waste can be derived from a first-order reaction; thus the organic carbon in the amount of waste decays exponentially with time. It also assumes that the biogas production is proportional to the organic carbon present in the waste and the produced biogas consists of 50% v/v methane. In the development of this model, the organic matter was assumed to be predominantly cellulose. Assuming that all cellulose is completely converted to biogas, the methane production per kg of organic carbon biodegraded is 0.933 m^3 (normalized to 1 atm and 0°C) corresponding with a biogas (LFG) production of 0.75 m^3 per kg OM degraded. (Mor et al. 2006; Scharff and Jacobs 2006). The amount of biodegradable carbon in the waste (cellulose) is 400 g/kg. Therefore the methane generation capacity per ton of waste (cellulose), assuming that all organic carbon is converted, is 400 * 0.933 (m^3 CH_4/kg C) = 373 m^3 CH_4/ ton waste. In the TNO model it is assumed that coarse household waste and household waste have an inorganic biodegradable carbon content of 130 kg C /ton wet waste. In that case the production of methane from this type of waste amounts 130* 0.933 (m^3 CH_4/kg C) = 121 m^3 CH_4/ ton waste, a value comparable to the LandGEM.

This finally results in a model in which the biogas production is proportional to:
$$\text{the amount of biodegradable organic carbon} * k_T * e^{-k_T * t}$$

Where;
k_T TNO reaction rate constant ($year^{-1}$)
t time elapsed since deposition (year)

The amount of biodegradable organic carbon is assumed to be proportional to the amount of biodegradable organic matter.

5.2.4 Multi-phase model (Afvalzorg)

The Afvalzorg-model was developed by NV Afvalzorg in the Netherlands. It is based on a combination of literature information (as accumulated in the 2006-IPCC model) and own experiences with landfill gas generation and measured emissions at the Afvalzorg-sites at Nauerna, Braambergen and Wieringermeer in the Netherlands (Scharff and Jacobs 2006). The Afvalzorg model is a multi-phase model that describes landfill gas generation as a function of biodegradable organic matter (OM) in deposited waste comprised of different fractions. The advantage of a multi-phase model is that typical waste composition can be taken into account. In the this model, the biodegradable fraction is split into three fractions with different biodegradation rate constants. For each fraction it is assumed that the rate of LFG production can be derived from a first-order degradation of the amount of organic matter of that fraction. Three biodegradation rate constants can be defined for slow, moderate and rapid biodegradation. This means that the production rate of biogas is proportional to the sum of the biodegradation rates of the three different fractions. It is further assumed in the Afvalzorg-model that the produced biogas contains 50% v/v methane. The rate constants used in the Afvalzorg multi-phase model are described in Table 5-2. The methane generation capacity per ton of waste assuming all organic matter (average amount) in the household waste is converted is 208 (kg OM/ton waste) * 0.72 m^3 (LFG/kg OM) = 237 m^3 LFG/ ton waste.

Table 5-2: Biodegradation rate constant k_A, organic matter content and conversion factor of the three biodegradable fractions of organic matter of household waste used in the Afvalzorg multi-phase model

Degradability	k_A (year^{-1})	Organic matter content (kg OM/ton waste)	Conversion factor (m^3 LFG/ kg OM)
Rapid	0.202	60-70	0.74
Moderate	0.105	75-90	0.72
Slow	0.030	45-48	0.7

Source: Adopted and modified from Scharff et al.(2006)

5.2.5 Other models

5.2.5.1 GasSim

GasSim is a multi-phase model developed by Golder Associates 2010 for the Environment Agency of England and Wales. GasSim multi-phase uses carbon content to calculate methane production and emission. GasSim is based on UK waste statistics and starts from hemicelluloses and cellulose content in the various waste fractions. For each waste fraction a biodegradable organic carbon is assumed. It quantifies all landfill gas related problems of a landfill, ranging from methane emissions, effects of utilization of landfill gas on local air quality to landfill gas migration via the subsoil to adjacent buildings.

The GasSim model version 1.00 of June 2002 is equipped with two approaches to calculate an estimate of methane emissions (GasSim manual Version 1.00). The first approach uses the GasSim multi-phase equation, which is based upon a multi-phase model described by Scheepers and van Zanten (1994) in Scharff et. al (2006). The second approach to estimate methane formation is the LandGEM model (Gregory et al. 2003a). The multi-phase model requires input of waste characteristics and the specific breakdown of waste categories and their carbon content during the particular year of disposal. The model is an executable and default values used, algorithms applied and assumptions made are protected in the program. Biodegradation rate constants k for dry, average wet and wet fractions and description of waste fractions translated from Afvalzorg waste categories of the GasSim model are presented in Table 5.3. According to Scharff et al.(2006) it is possible to include extraction efficiency of the LFG recovery system in the model and let GasSim calculate total surface emissions.

Table 5-3: Biodegradation rate constant, k values and waste fractions of the GasSim multi-phase model

Degradability	**k values** (year^{-1})			Fraction
	Dry	Average	Wet	
Rapid	0.076	0.116	0.694	Putrescibles, fines, garden wastes, sewage sludge, incinerator ash
Moderate	0.046	0.076	0.116	1/4 paper (excluding newspaper), nappies, miscellaneous combustible, composted organic material
Slow	0.013	0.046	0.076	3/4 paper (excluding newspaper), newspaper, textiles

Source: Scharff et al.(2006)

However, this feature in GasSim only functions if waste in place is capped to a certain degree. This can be activated in the model by checking a checkbox and giving a percentage of waste capped. If the checkbox is not checked, but a recovery is operational and the efficiency is given in GasSim, the model does not take extraction efficiency of the LFG recovery system into account. The latest versions available are a commercial version GasSim 2.1 and freeware version is GasSim lite 1.5 (Scharff and Jacobs 2006).

5.2.5.2 French E-PRTR

The French E-PRTR-model is a modified first-order decay model. The model output is methane generation of 4.8 kg (6.6 m^3) per ton waste per year in the first 5 years after landfilling; 2.4 kg per ton waste per year the 5 years after, 1.3 kg per ton waste per year in the 2nd decade and 0.6 kg per ton waste per year in the 3rd decade after landfilling. For moderately decomposable waste (e.g. non-hazardous industrial waste; household waste that is milled or composted), methane formation is assumed to be 50% of these values. The model is not available as a spreadsheet, but consists of a simple fill-in table (Oonk 2010).

5.2.5.3 EPER model France

The French EPER model gives two approaches to estimate methane emissions from landfills whereby the landfill operator has the choice to select the approach. The two approaches are:

a) estimates for landfill cells connected to an LFG recovery system using data of recovered LFG and the recovery efficiency;

b) estimates for landfill cells connected or not connected to an LFG recovery system using a multi-phase model (ADEME model) and the LFG recovery efficiency.

In the second approach, the French model mentions three fractions and three k values for each waste category. The three categories are similar to the fractions used in the Afvalzorg model (Scharff and Jacobs 2006).

The model describes three categories of waste and every category has a specific methane generation capacity per ton of waste. In the context of this thesis, only the specific methane generation capacity and k_L values for MSW, sludges and yard waste (also referred to as household waste, sewage sludge and compost in the Afvalzorg model) are given consideration. K_L values and the respective fraction distribution are presented in Table 5.4.

Table 5-4: Waste fractions and k values used in the ADEME multi-phase model

Degradation rate	**Percentage** (%)	$\mathbf{k_L}$ (year^{-1})
Rapid	15	0.50
Moderate	55	0.10
Slow	30	0.04

Adopted and modified from Scharff et al.(2006)

The waste has an initial methane generation capacity of 100 m^3 methane per ton of waste. Like all previously mentioned models, it also assumes that the produced biogas is 50% v/v methane.

Summary of the models

The various models are in general dealing with aspects such as the waste deposited, landfill gas (biogas) production rate, conversion or assimilation factors, rate constants and time. However, the models show also some differences, although this is not always fully clear from literature, because essential information is sometimes missing. In Table 5-5 an attempt has been made to compare the various models.

Table 5-5: Summary of variables as used by variable models

Category	Input parameter	PRIMARY MODELS			OTHER MODELS		
		LandGem	**TNO**	**Afvalzorg Multiphase**	**GasSim**	**French E-PRTR**	**EPER France**
Waste	amount of biodegradable organic carbon		✓		✓		
	amount of waste deposited			✓		✓	
	amount of biodegradable organic matter			✓			
	different waste fractions			✓	✓		✓
	amount of waste in a section	✓					
Landfill gas	LFG production at any given time		✓	✓			
	methane generation potential	✓	✓			✓	✓
	annual methane production rate	✓			✓		✓
Factor	dissimilation factor		✓	✓			
	conversion factor of biodegradable organic Carbon into LFG		✓				
	conversion factor of OM into LFG			✓			
	normalisation factor						✓
Constant	LFG production rate constant		✓	✓	✓		✓
	methane production rate constant	✓					
Time	age of waste since deposition		✓	✓			✓
	age of the various sections since deposition of waste	✓			✓		

All the models described in this chapter use first-order reaction mechanisms and they use the following variables in their calculations:

- ***Waste***: amount of biodegraded organic carbon; amount of waste deposited/landfilled; amount of organic matter; different waste fractions; mass of waste in a section;
- ***LFG***: LFG production rate; methane generation potential; annual methane production rate;
- ***Factors***: dissimilation factor; conversion factor of either Carbon or organic matter OM into LFG; normalization factor;
- ***Constants***: LFG or methane production rate constant;
- ***Time***: average age of waste since deposition; age of the section with deposited waste.

5.3 Modeling of LFG

This section presents two developed modified models of methane generation in LFB and some theoretical calculations for one cell and many cells landfilled over a specified period of time. The modified models are derived from the LandGem model described by U.S.EPA and Scharff et al.(2006) and the multi-phase model which are both highly recommended by the US Environmental Protection Agency (USEPA 1998, 2005b) and the Intergovernmental Panel on Climate Change (IPCC 2006). These models both start with the assumption that waste stabilization in LFBs can be described well as a first-order decay process. They are generally recognized as being the most widely used approaches.

5.3.1 Modified single - phase model

Waste deposited in a LFB is comprised of organic and inorganic matter. Not all organic matter present can be converted to LFG and in practice not every fraction that can be converted microbiologically will be converted, mainly because environmental conditions in certain parts of the waste disposal site inhibit biological activity. It is assumed that waste is filled in a cell and stays in the cell for the same length of time before the cell is closed (i.e. all waste in a cell has the same age). The total amount of organic matter (OM) that is biodegradable can be formulated as the product of the total amount of waste deposited and the average fraction of biodegradable organic matter that can converted into LFG. The total amount of organic matter at a certain time t in the waste is then given as:

$$OM = M^* X \qquad (5 - 1)$$

Where;
M = total amount of waste present in the LFB at time t (ton waste);
X = fraction of biodegradable organic matter in the waste at time t (kg OM/ton waste).

The initial total amount of waste disposed at t = 0 is M_o and the biodegradable fraction at t = 0 is X_o. This means that the total amount of organic matter at t = 0 is $M_o{}^* X_o$.

The LFG potential is based on the amount of organic matter in the site corrected for the amount of organic matter that does not biodegrade under the conditions of the landfill.

Models for biodegradation of the organic matter and LFG (biogas) generation are in general based on a first-order biodegradation process in the amount of OM. According to this first-order biodegradation process, LFG and consequently methane is being formed immediately after deposition of the waste and gradually being reduced in time. The production rate of LFG can be described by equation (5-2):

$$q = M * X * k \qquad (5 - 2)$$

Where:
q = LFG production rate from biodegradable organic fraction in the waste at time t (kg/year);
k = first-order rate constant for LFG production ($year^{-1}$);
M = the total amount of waste present at time t (ton);
X = fraction of biodegradable organic matter in the waste at time t (kg OM/ton waste).

The amount of organic matter and also the total amount of waste is reduced as LFG is produced. The relationship between the production rate of LFG (q) and the total amount of waste reduced is expressed by equation (5-3):

$$q = -\zeta * \frac{d(M * X)}{dt} \ldots\ldots\ldots\ldots\ldots\ldots (5-3)$$

Where:
ζ = correction factor;
t = time (years).

The factor ζ gives a correction for the biological conversion process of organic matter to LFG whereby the reduction in total mass is in general not equal to the mass of LFG that is produced. In general this factor results into a value that can be somewhat lower or higher than 1. There are several causes for this phenomenon:

- Part of the organic matter is used for growth of micro-organisms that are responsible for the conversion process and is not converted in biogas;
- Biogas contains only CH_4 and CO_2. Organic matter that is biodegraded in biogas contains not only the elements C, O and H but also small amounts of the elements N and S. In the conversion process these elements are not converted in biogas;
- The ratios between C, O and H not always fit in the conversion process to a mixture of CH_4 and CO_2 only. Often a net amount of water is produced, often a net amount water is necessary.

From literature it is not always clear at what conditions of temperature and pressure the amount of LFG is defined. In this thesis we assume that the gas is always at STP (1 atmosphere and 0^o C) and a composition of 50% methane and 50% carbon dioxide. Then the density is approximately 1.35 kg/m^3. This means that the correction factor due to the transformation from kg into m^3 is somewhat higher than 1.
The first type of organic matter we will discuss is cellulose. The composition of cellulose is given by the formula: $C_6H_{12}O_6$. The biodegradation of cellulose into biogas is described by equation (5-4a). All the organic matter in the cellulose is converted to biogas. The biogas consists of 50% methane and 50% carbon dioxide on a volume basis.
For other types of organic matter the conversion efficiency and also the composition of the obtained biogas is different. For example for the substance Antamanide the composition is given by the formula: $C_{64}H_{78}O_{10}N_{10}$. The biodegradation of Antamanide in biogas is given by equation (5-4b). As can be seen from the equation of the conversion process also water is used and ammonia is produced. Here, the volume ratio of methane and carbon dioxide is not exactly 50/50. In general the biogas potential of organic matter is given by the Buswell equation (Buswell and Neave 1930). This Buswell equation is given by equation (5-4c). The potential content of biogas of a certain organic compound can be calculated if the chemical composition of the compound is known.

$$C_6H_{12}O_6 \rightarrow 3CH_4 + 3CO_2 \ldots\ldots\ldots\ldots\ldots\ldots (5-4a)$$

$$C_{64}H_{78}O_{10}N_{10} + 47H_2O \rightarrow 35.5CH_4 + 28.5CO_2 + 10NH_3 \ldots\ldots\ldots\ldots (5-4b)$$

$$C_aH_bO_cN_d + \left(\frac{4a-b-2c+3d}{4}\right)H_2O \rightarrow \left(\frac{4a+b-2c-3d}{8}\right)CH_4 + \left(\frac{4a-b+2c+3d}{8}\right)CO_2 + dNH_3 \ldots\ldots (5-4c)$$

The composition of organic matter in municipal solid waste (MSW) is very complex, varies strongly between the various biodegradable compounds of MSW and is often difficult to measure. Because the correction factor ζ deviates not very strongly from 1 we make the assumption that the correction factor $\zeta = 1$.

Substituting equations (5-2) in (5-3) results in:

$$k * (M * X) = -\frac{d(M * X)}{dt} \ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots (5-5)$$

The initial condition of equation (5-5) is;

At time $t = 0$:
$M = M_o$ and $X = X_o$

Where:
M_o = initial amount of waste deposited (ton)
X_o = initial fraction of biodegradable organic matter in the waste (kg OM/ton waste)

Integration of equation (5-5) results into

$$M * X = M_o * X_o * e^{-k*t} \ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots (5-6)$$

Substitution of equation (5-2) into (5-6) results in

$$q = M_o * X_o * k * e^{-k*t} \ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots (5-7)$$

Therefore the LFG production rate is given by:

$$q = M_o * X_o * f * k * e^{-k*t} \ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots (5-8)$$

The total amount of LFG, Q, that is produced from the waste after a time t, is given by:

$$Q = \int_{t=0}^{t=t} q\, dt = M_o * X_o * f * k * \int_{t=0}^{t=t} e^{-k*t}\, dt \ldots\ldots\ldots\ldots\ldots\ldots (5-9)$$

Or

$$Q = M_o * X_o * f * \left(1 - e^{-k*t}\right) \ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots (5-10)$$

Where:
Q total amount of LFG (m^3);
M_o initial amount of waste deposited (ton);
X_o initial biodegradable organic matter fraction in the waste (kg OM/ton waste);
k adapted first order reaction rate constant ($year^{-1}$);
t time elapsed since deposition (year);
f conversion factor (m^3LFG/kg OM converted).

If we assume that the biogas has a composition of 50% methane and 50% carbon dioxide, then the density of this biogas at standard conditions is approximately 1.35 kg/m^3. If all biodegradable organic matter is converted in biogas, then $f = 1/1.35=0.75$

5.3.2 Modified multiphase model

The organic matter in the waste (M * X), consists of biodegradable and non-biodegradable organic materials. In general the biodegradable part is not fully converted to LFG during the disposal time. The biodegradable organic part of waste also does not consist of a single component, but of a spectrum of components with different biodegradability. It is assumed that the biodegradable organic matter can be subdivided into three fractions with different biodegradability. The subdivision results in a fraction that slowly biodegrades ($X_{o,s}$), a fraction that biodegrades moderately ($X_{o,m}$) and a rapidly biodegradable fraction ($X_{o,r}$). The total production rate of biogas is therefore calculated by taking the summation of the three production rates. The resulting expression is a first-order model comparable with the multi-phase model (Afvalzorg model).

Including the subdivisions of the various types of biodegradable organic matter with different reaction rate constants, the LFG production rate, q, is given by:

$$q = \mathrm{M_o} * \left(\mathrm{X_{o,s}} * \mathrm{f}_s * \mathrm{k_s} * \mathrm{e}^{-\mathrm{k_s}*\mathrm{t}} + \mathrm{X_{o,m}} * \mathrm{f}_m * \mathrm{k_m} * \mathrm{e}^{-\mathrm{k_m}*\mathrm{t}} + \mathrm{X_{o,r}} * \mathrm{f}_r * \mathrm{k_r} * \mathrm{e}^{-\mathrm{k_r}*\mathrm{t}}\right) \ldots\ldots\ldots\ldots\ldots\ldots (5-11)$$

and the total amount of LFG, Q, that is produced from the waste after a time t, is given by

$$\mathrm{Q} = \mathrm{M_o} * \left(\mathrm{X_{o,s}} * \mathrm{f_s} * (1 - \mathrm{e}^{-\mathrm{k_s}*\mathrm{t}}) + \mathrm{X_{o,m}} * \mathrm{f_m} * (1 - \mathrm{e}^{-\mathrm{k_m}*\mathrm{t}}) + \mathrm{X_{o,r}} * \mathrm{f_r} * (1 - \mathrm{e}^{-\mathrm{k_r}*\mathrm{t}})\right) \ldots\ldots\ldots\ldots\ldots\ldots (5-12)$$

Note: $X_{o,s} + X_{o,m} + X_{o,r} = X_o$.

Where:

M_o	initial amount of waste deposited (ton);
$X_{o,s}$	initial fraction of organic matter slowly biodegradable (kg OM/ton waste);
$X_{o,m}$	initial fraction of organic matter moderately biodegradable (kg OM/ton waste);
$X_{o,r}$	initial fraction of organic matter rapidly biodegradable (kg OM/ton waste);
f_s	conversion factor of slowly biodegradable organic matter (m^3LFG/kg OM);
f_m	conversion factor of moderately biodegradable organic matter (m^3LFG/kg OM);
f_r	conversion factor of rapidly biodegradable organic matter (m^3LFG/kg OM);
k_s	adapted first-order reaction rate constant for slowly biodegradable waste ($year^{-1}$);
k_m	adapted first-order reaction rate constant for moderately biodegradable waste ($year^{-1}$);
k_r	adapted first-order reaction rate constant for rapidly biodegradable waste ($year^{-1}$);
t	time elapsed since deposition (year).

In various literature sources the biodegradation rate constants for different fractions of organic matter range from 0.01 to 0.7 (yr^{-1}). For the purpose of demonstrating the effect of the modified model equation (5-12), a plot of the efficiency of LFG production versus time is made. The plot applies the biodegradation rate constants of 0.02, 0.2 and 0.4 (yr^{-1}) of the three fractions (slowly, moderately and rapidly biodegradable substances respectively). The expression for the conversion of a certain fraction is given by equation(5-13):

$$\frac{Q}{Q_{max}} = 1 - e^{-k*t} \ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots (5-13)$$

Q total amount of LFG (m^3);
Q_{max} maximum amount of LFG that can be produced at time t (m^3);

From Figure 5-1, several deductions can be drawn. For instance, it is expected that 10%, 63% and 86% of the slow, moderate and rapid biodegradable waste respectively shall be converted into LFG after 5 years. It will take another 5 years to achieve additional 10%, 30% and 14% conversion of slow, moderate and rapid biodegradable waste respectively to LFG. It can therefore be said that 10 years is the selected optimal operation time of active cells for the adapted LandGEM model as discussed in the subsequent section. A duration of 10 years is selected because at that time a significant amount of the biodegradable organic matter will be converted to LFG.

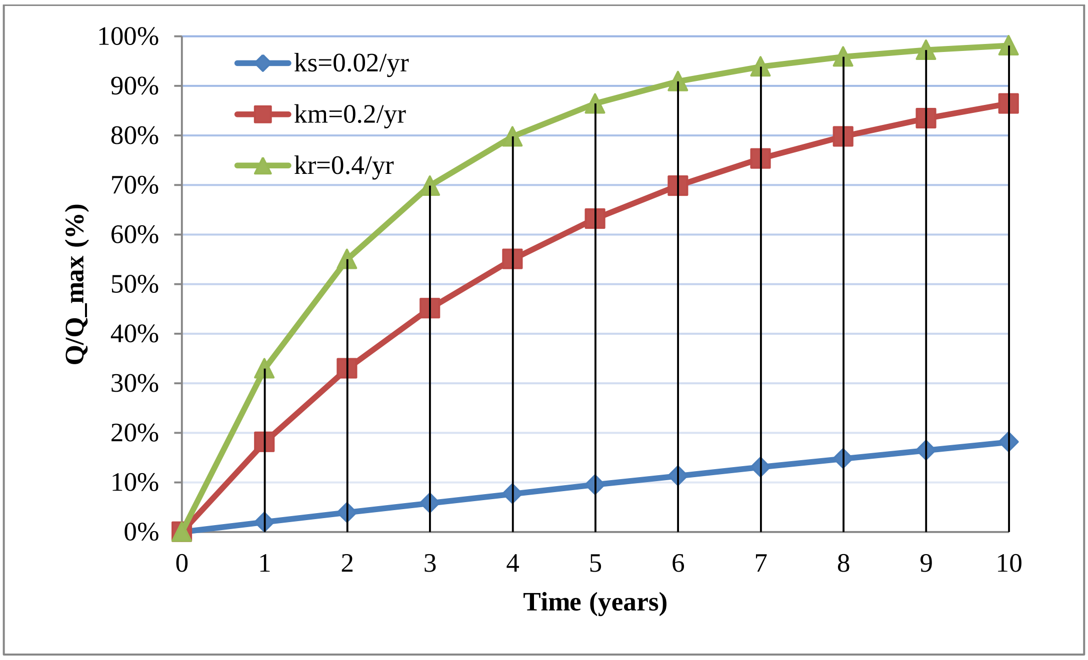


Figure 5-1: Efficiency of LFG production of the slowly, moderately and rapidly biodegradable organic matter as a function of time

Figure 5-2 deduced from equation (5-12) gives the efficiency of LFG production versus time on a non-linear axis applying three different degradation rate constants.

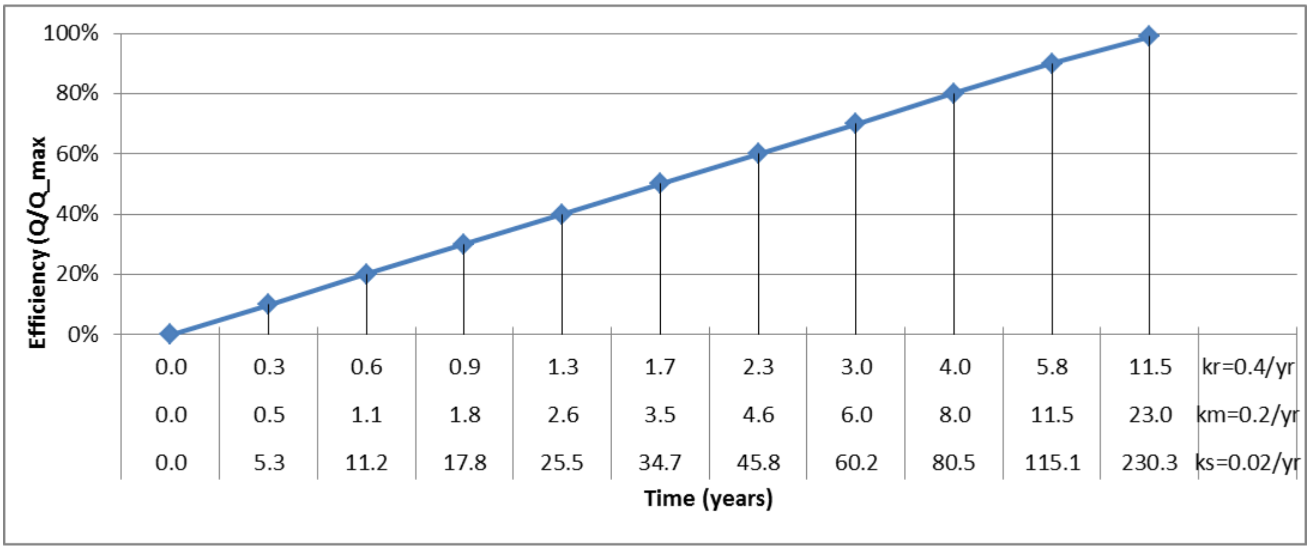


Figure 5-2: Efficiency of LFG production of the slowly, moderately and rapidly biodegradable organic matter against time plotted on a non-linear graph

In Figure 5-2, the difference between the degradation rate constants is 10-fold between k_m and k_s whereas between k_r and k_m it is double. The impact can be seen in Figure 5-2. Taking the example of 80% efficiency, it will take 4 years for the rapidly biodegradable and double that time for the moderately and twenty times for the slowly biodegradable substances to be converted to LFG.

5.3.3 Adapted LandGEM

This modified model is built to suit a number of N cells of a landfill site filled with an equal amount of biodegradable waste. It is assumed that one cell is filled per week (i.e. the time taken to fill N cells is $^N/_{52}$ (year) and that after this period we continue with this process during a period of 10 years. After 10 years when the 521st cell becomes active, the 1st cell is taken out of operation (no collection of biogas anymore) and from that time on we take one cell out of operation (the oldest one at that time) each time a new cell is opened and becomes active in the biogas production process. Therefore at any time, only 520 cells are active in biogas production and we get a stationary situation regarding the production of biogas (See Figure 5-3).

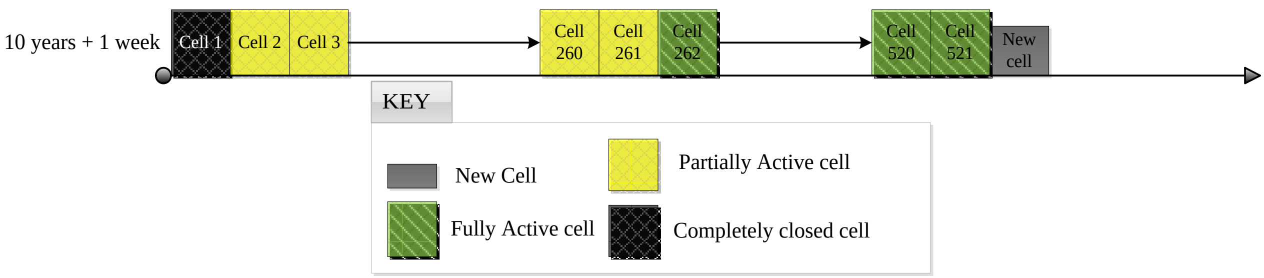


Figure 5-3: Filling, activation and closure of cells

Suppose we start at t = 0 with one cell and continue filling each week one new cell, then the total LFG production at time t, after N cells have been filled and become active, is given by:

Cell 1: $q_1 = M_{oc} * X_o * f * k * e^{-k*t}$

Cell 2: $q_2 = M_{oc} * X_o * f * k * e^{-k*\left(t-\frac{1}{52}\right)}$

Cell 3: $q_3 = M_{oc} * X_o * f * k * e^{-k*\left(t-\frac{2}{52}\right)}$

•

•

•

Cell N: $q_N = M_{oc} * X_o * f * k * e^{-k*\left(t-\frac{N-1}{52}\right)}$

Therefore the overall LFG production rate after filling N cells is given by:

$$q_{total} = \sum (q_1 + q_2 + q_3 + \cdots \ldots \ldots q_N) \ldots \ldots \ldots \ldots \ldots \ldots \ldots \ldots \ldots \ldots (5-14)$$

$$q_{total} = \begin{pmatrix} M_{oc} * X_o * f * k * e^{-k*t} & + \ M_{oc} * X_o * f * k * e^{-k*\left(t-\frac{1}{52}\right)} \ + \\ M_{oc} * X_o * f * k * e^{-k*\left(t-\frac{2}{52}\right)} & + \cdots M_{oc} * X_o * f * k * e^{-k*\left(t-\frac{N-1}{52}\right)} \end{pmatrix} \ldots \ldots (5-15)$$

$$= M_{oc} * X_o * \ f * k * \left(e^{-k*t} \ + \ e^{-k*\left(t-\frac{1}{52}\right)} \ + \ e^{-k*\left(t-\frac{2}{52}\right)} + \cdots \ldots \ldots e^{-k*\left(t-\frac{N-1}{52}\right)}\right)$$

$$= \ M_{oc} * X_o * f * k * e^{-k*t}\left(1 \ + \ e^{\left(\frac{k}{52}\right)} \ + \ e^{\left(\frac{2k}{52}\right)} + \cdots \ldots \ldots \ldots \ldots \ldots e^{\left(\frac{(N-1)k}{52}\right)}\right)$$

•

•

•

$$q_{total} = M_{oc} * X_o * \ f * k * e^{-k*t}\left(\frac{1 - e^{\left(\frac{N*k}{52}\right)}}{1 - e^{\left(\frac{k}{52}\right)}}\right) \ldots \ldots \ldots \ldots \ldots \ldots \ldots \ldots (5-16)$$

Equation (5-16) is valid for $t \geq {}^{N}/_{52}$, where:

q_{total}	total biogas production rate from the N cells (m^3/year);
M_{oc}	initial amount of waste deposited in the cell (ton);
X_o	initial fraction of biodegradable organic matter in the waste (kg OM/ton waste);
f	conversion factor (m^3 LFG/kg OM converted);
k	first order reaction rate constant ($year^{-1}$);
N	number of cells filled.

Therefore the adapted LandGEM becomes a modified model (equation 5-16) dependent on the number of cells filled, the time t and the initial mass of biodegradable organic matter deposited in the cells.

For a fixed number of N active cells and one cell filled and one cell closed (the oldest one at that time) per week we get a stationary situation. Then the total LFG generation rate from these N cells is given by substitution of $t = {}^{N}/_{52}$ in equation (5-16). This equation can be re-written as:

$$q_{total} = M_{oc} * X_o * f * k * \left[e^{-k*t} * \left(\frac{1 - e^{k*t}}{1 - e^{\left(\frac{k}{52}\right)}}\right)\right] \ldots \ldots \ldots \ldots \ldots \ldots \ldots \ldots (5-17)$$

or:

$$q_{total} = M_{oc} * X_o * f * k * \left(\frac{e^{-k*t} - 1}{1 - e^{\left(\frac{k}{52}\right)}} \right)$$

For k << 52 we get:

$$q_{total} = M_{oc} * X_o * f * k * \left(\frac{e^{-k*t} - 1}{-\frac{k}{52}} \right)$$

or:

$$q_{total} = 52 * M_{oc} * X_o * f * (1 - e^{-k*t}) \dots\dots\dots\dots\dots\dots\dots\dots\dots.\dots\dots\dots\dots (5-18)$$

We can calculate the total amount of biogas produced per year also more directly. In one year a total amount of organic waste of $52*M_{oc}*X_o$ is disposed in the landfill. After 10 years these cells are taken out of operation and closed. The amount of biogas that is produced from these amount of organics in 10 years is given by substitution of t = 10 in equation (5-10)0. In the stationary situation, in which each year a total amount of organic waste of $52M_{oc}*X_o$ is disposed in the landfill, then this amount corresponds to the total annual production rate of biogas from the site.

For a mixture of fractions with slow, moderate and high biodegradability the biogas production rate is given by:

$$q_{total} = M_{oc} * \begin{pmatrix} X_{o,s} * \ f_s * k_s * e^{-k_s*t} \left(\frac{1 - e^{\left(\frac{N*k_s}{52}\right)}}{1 - e^{\left(\frac{k_s}{52}\right)}} \right) \\ + \\ X_{o,m} * \ f_m * k_m * e^{-k_m*t} \left(\frac{1 - e^{\left(\frac{N*k_m}{52}\right)}}{1 - e^{\left(\frac{k_m}{52}\right)}} \right) \\ + \\ X_{o,r} * \ f_r * \ k_r * e^{-k_r*t} \left(\frac{1 - e^{\left(\frac{N*k_r}{52}\right)}}{1 - e^{\left(\frac{k_r}{52}\right)}} \right) \end{pmatrix} \dots\dots\dots\dots\dots\dots..\dots (5-19)$$

Where:

q_{total} total biogas production rate after filling N cells (m^3/year);
M_{oc} initial amount of waste deposited in the cell (ton);
$X_{o,s}$ initial fraction of organic matter slowly biodegradable (kg OM/ton waste);
$X_{o,m}$ initial fraction of organic matter moderately biodegradable (kg OM/ton waste);
$X_{o,r}$ initial fraction of organic matter rapidly biodegradable (kg OM/ton waste);
f_s conversion factor of slowly biodegradable organic matter (m^3 LFG/kg OM);
f_m conversion factor of moderately biodegradable organic matter (m^3LFG/kg OM);
f_r conversion factor of rapidly biodegradable organic matter (m^3 LFG/kg OM);
k_s first-order reaction rate constant for slowly biodegradable waste ($year^{-1}$);
k_m first-order reaction rate constant for moderately biodegradable waste ($year^{-1}$);

k_r first-order reaction rate constant for rapidly biodegradable waste ($year^{-1}$);
t time elapsed since deposition (year).

5.4 Conclusions

1. There exist several models in various literature sources dealing with the production of biogas from landfills and landfill bioreactors. All these models may assist in the determination of landfill gas production rate and the potential amount of biogas produced but they do not consistently use the same input data.

2. The production of biogas or methane is described in literature as a first-order reaction (with reaction rate constant k) in the total amount of waste (wet waste), the total amount of biodegradable organics, or the amount of bioavailable organic carbon. Some models include differences in the bioavailability of the various types of organic material present in a waste. Some models also include the effect of differences in the age (lifetime) of the disposed material.

3. Based on the different models as mentioned in literature an innovative model has been derived. This model is also a first-order biodegradation model based on the conversion of three different types of biodegradable organic matter. It presents the production of biogas as a function of the composition of the waste, especially with respect to the biodegradable organic matter and the reaction rate constants of these three types of organics. With the model it is possible to calculate the production of biogas as a function of time and the composition of the waste.

4. From literature a lot of data required for the calculation of the biogas production from waste in a landfill or landfill bioreactor have been derived. These data deals with the amount of organic material in the waste, the different fractions of organic material in the waste, the total solids, the potential amounts of biogas in the waste, and the values of the various reaction rate constants

CHAPTER 6

Pilot-scale experiment on anaerobic landfill bioreactor in Tanzania

6.1 Introduction

Landfill bioreactors have been developed in the U.S.A, the EU, Australia, Japan, and other countries, as a long-term municipal solid waste (MSW) management option (Reinhart et al. 2002). Although their long-term performance has yet to be fully understood, there are many advantages to the operation of landfills as bioreactors which include (1) increased potential for waste to energy conversion by improving the LFG generation rate, (2) storage and/or partial treatment of leachate, (3) increased landfill capacity due to enhanced settlement and increased air space, and (4) reduced waste decomposition time from several decades to 5–10 years thus reduced land use costs.

Some researchers have conducted studies on Landfill bioreactors (LFB) with characteristics of waste almost similar to that of East Africa. San and Onay (2001) filled a simulated recirculated landfill reactor with a synthetic mix of MSW whose characteristics were mostly food waste being 76% on dry basis. The initial moisture content of the waste was 80%. They researched the impact of various leachate recirculation regimes on waste degradation in landfills and in situ leachate treatment to provide data for successful operation of landfill sites in the Istanbul metropolitan area. This study showed that landfill leachate management with leachate recirculation is a promising and challenging strategy. Leachate recirculation is a feasible way for in situ leachate treatment decreasing the cost of further external treatment. Reintroduction of necessary nutrients such as phosphorus and nitrogen, enhanced the growth of microbial population and the extent of stabilization (San and Onay 2001).

Sponza and Ağdağ (2004) evaluated the impact of leachate recirculation and recirculation volume on stabilization of municipal solid wastes in simulated anaerobic bioreactors. Their waste had a moisture content of 75-86% and organic waste constituting 75-90% of the wet waste. The results of this study showed that the COD and VFA concentrations in leachate were very high but an optimum leachate recirculation volume contributes to enhanced COD removal, decreasing VFA, and effective methane gas production. The results also showed the feasibility of leachate recirculation in reducing the overall leachate generation for treatment and in enhancing the degradation of solid waste.

Another study was a pilot-scale experiment on anaerobic bioreactor landfills in China conducted by Jiang et al. (2007) using fresh waste unloaded from daily municipal waste collection trucks with physical properties of 60% moisture content (wet weight basis) and 75% volatile solids (VS, dry basis). Findings from this study have shown that leachate recirculation with a high rate such as 213 mm/day can be adopted as an effective in situ pre-treatment approach to remove organic pollutants in leachate and notably ammonia-nitrogen, phosphorus and some persistent organic compounds can be accumulated in the effluent leachate that need further treatment.

Despite these promising studies, it appears that there is limited data on the performance of LFBs in tropical developing countries. In East Africa no landfills have yet been designed and operated as recirculated landfills. It is important to better understand the possibilities of LFBs in situations with waste having a high organic matter and moisture content and a high temperature.

In this chapter, findings from a comparative study of a pilot scale landfill bioreactor and sanitary landfill are presented. This pilot scale experiment was conducted in Dar es Salaam city, Tanzania, East Africa, to study the effect of recirculation on waste degradation and acidification, landfill gas production, and in situ leachate treatment. In order to achieve this objective the following activities were undertaken: (1) study of the variations of the effluent leachate characteristics as an indicator of waste stabilization, (2) evaluation of the effects of leachate recirculation on leachate COD removal, (3) evaluation of the landfill gas generation rate and composition, (4) monitoring of the settlement of waste due to the organic matter degradation.(5) investigation of the possibility to keep the landfill acidified during a certain period

6.2 Materials and methods

In this section the pilot reactors, the loading protocol, the characteristics of the waste and analytical procedures are described.

6.2.1 Description of the pilot-scale LFB setup

The pilot setup consists of two reactors without (R1) and with (R2) leachate recirculation. R1 is operated as a control reactor simulating a sanitary landfill and R2 is considered as a simulated landfill bioreactor. The reactors are both built in a concrete structure with a square horizontal cross section of 1 by 1 m and 2.5 m deep to represent a landfill cell. The depth of the cell is in conformity with the landfilling requirements according to the excavated cell method recommended by Tchobanoglous et al. (1993, p. 374) whereby the cell depth ranges from 3 to 10 ft (0.9 to 3 m). Practically a landfill site consists of such cells one deposited on top of the other to a height of about 10 to 15 m.

Figure 6-1 shows the schematic representation of the experimental setup showing waste in reactors, leachate collection, leachate recirculation system comprised of a storage tank and small diameter leachate inlet pipes, LFG extraction pipes and temperature monitoring ports at different heights. Leachate recirculation pipes were laid in R2 only to recirculate leachate. At the bottom of both reactors, there is an outlet for the generated leachate to be collected. The bottom of the reactor was slightly slanted to direct the generated leachate towards the outlet and at the outlet a wire mesh was placed to prevent the waste from being carried out with the leachate.

Installation of LFG extraction pipes was done during the filling of the reactors with waste. Between the waste and the LFG pipes, 25-35 mm granite - gravel media were placed as drainage layers for leachate in downward direction and landfill gas in upward direction. The gravel was held in place by the aid of a PVC casing whereby the LFG pipes were enclosed between the casing and gravel. After the casing was fully surrounded with waste it was removed to leave the gravel in contact with the waste while shielding the LFG extraction pipes from direct contact with the waste. The waste was capped by compacted clay soil in which the leachate distribution system was laid with leachate inlet pipe running approximately 100 mm into the waste.

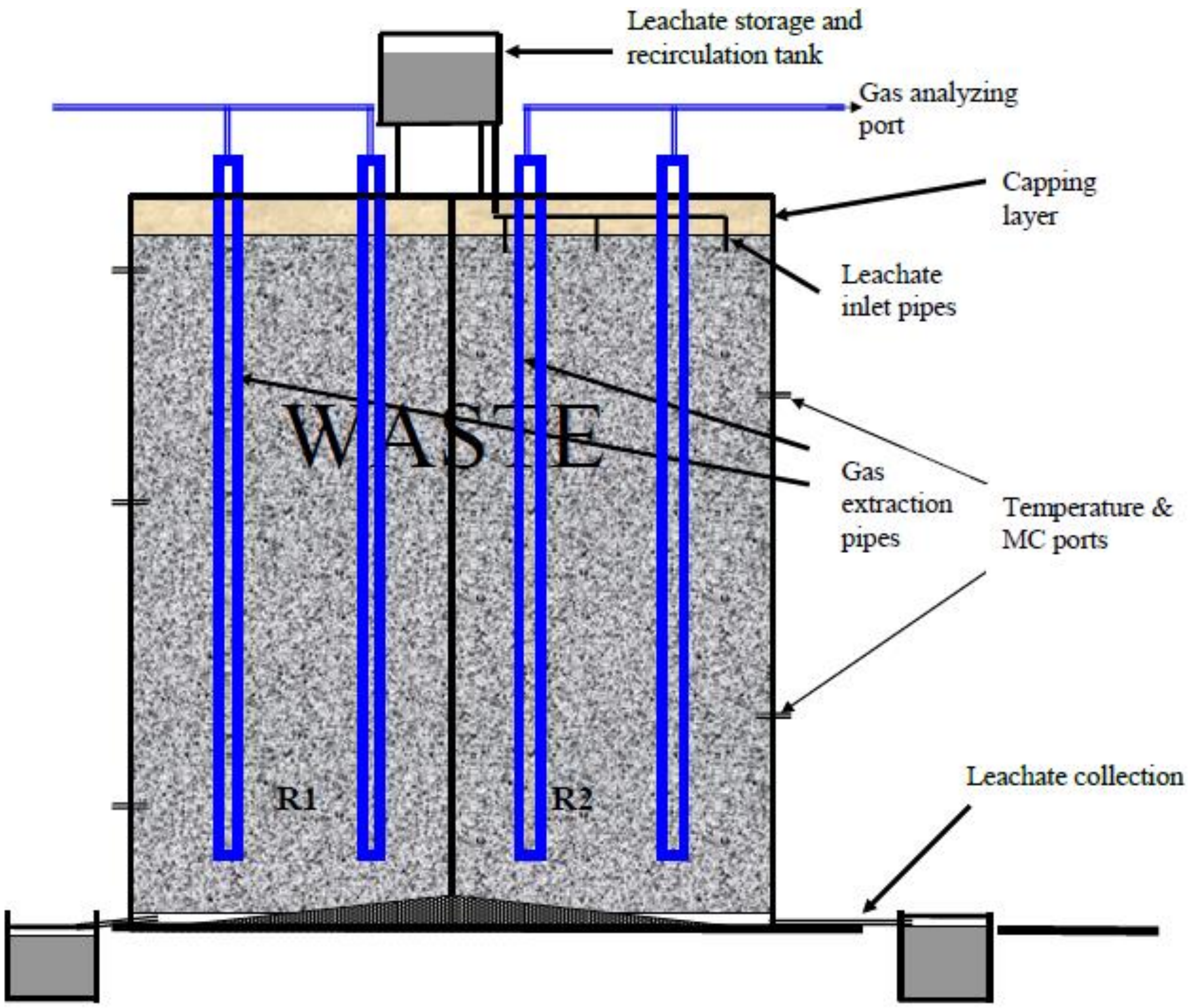


Figure 6-1: Schematic representation of the LFB

6.2.2 Loading and operation protocol

Each of the reactors were simultaneously filled with about 2.3 tons of wet waste of moisture content about 64%. The waste used was collected for several days from municipal waste transfer stations in Mwenge area in Kinondoni municipality, Dar es Salaam city. The collected waste was predominantly food waste (about 60%) and was sorted to remove glass, metals, plastics and any other non-biodegradable materials. The relatively high proportion of organic waste is considered to be a characteristic of MSW in Tanzania, as well as several other developing countries (San and Onay 2001; Sponza and Ağdağ 2004; Mbuligwe et al. 2002; Kassim and Ali 2006; Jiang et al. 2007). The sorted waste was loaded into the reactors and compacted manually using a sledge hammer to a density of nearly 900 ton/m^3. Table 6-1 shows the loading protocol of the LFBs.

Leachate from R1 was collected and samples for measurement were drawn but not included in the recirculation leachate. After drawing samples for measurement, all the remaining leachate from R2 was manually transferred to the storage tank at the top of the reactors and allowed to flow back into R2.

Table 6-1: Loading protocol of the landfill bioreactors

	Reactor without Recirculation R1	Reactor with Recirculation R2
Quantity of waste (kg)	2394	2342
Recirculation	Without	With
Recirculation rate (mm/day)	-	13 (average)
Moisture content (%)	64.07	64.38
Operation time (day)	378	365

Initially the waste was kept in a static state for five days before recirculation of leachate began in R2 at an average rate of 13 mm/day. At this rate the HRT of recirculated leachate in R2 would be 230 days if plug flow would be assumed. The actual HRT of the recirculated liquid is not known but probably less with possible short circuiting taking place.

Throughout the study of 52 weeks R1 was run as a flow-through system. The study with R2 was broken into two phases. During phase one the leachate was recirculated directly to the top of the reactor. COD, pH, temperature, conductivity, nutrients, volume of leachate generated, waste settlement and gas production were monitored. After 120 days of leachate recirculation in R2 the pH was observed to be too low for methanogenesis. Then phase two of the study began which involved recirculation of leachate after treatment via an Up-flow Anaerobic Sludge Blanket (UASB) reactor as an in-situ pre-treatment measure of the leachate. The UASB reactor used was a 15.7 litre PVC reactor of 2 m height, 0.1 m diameter, HRT of 1.15 days, filled with 6.75 L anaerobic sludge obtained from an existing UASB reactor whose sludge age is more than 5 years with a specific methanogenic activity of about 0.17 g COD/gVSS/day. Monitoring of the same parameters as for phase one were continued to be monitored for both R1 and R2. Figure 6-2 depicts the flow diagram showing the pilot scale experiment during the two phases and the two distinct regimes of leachate flow of R2.

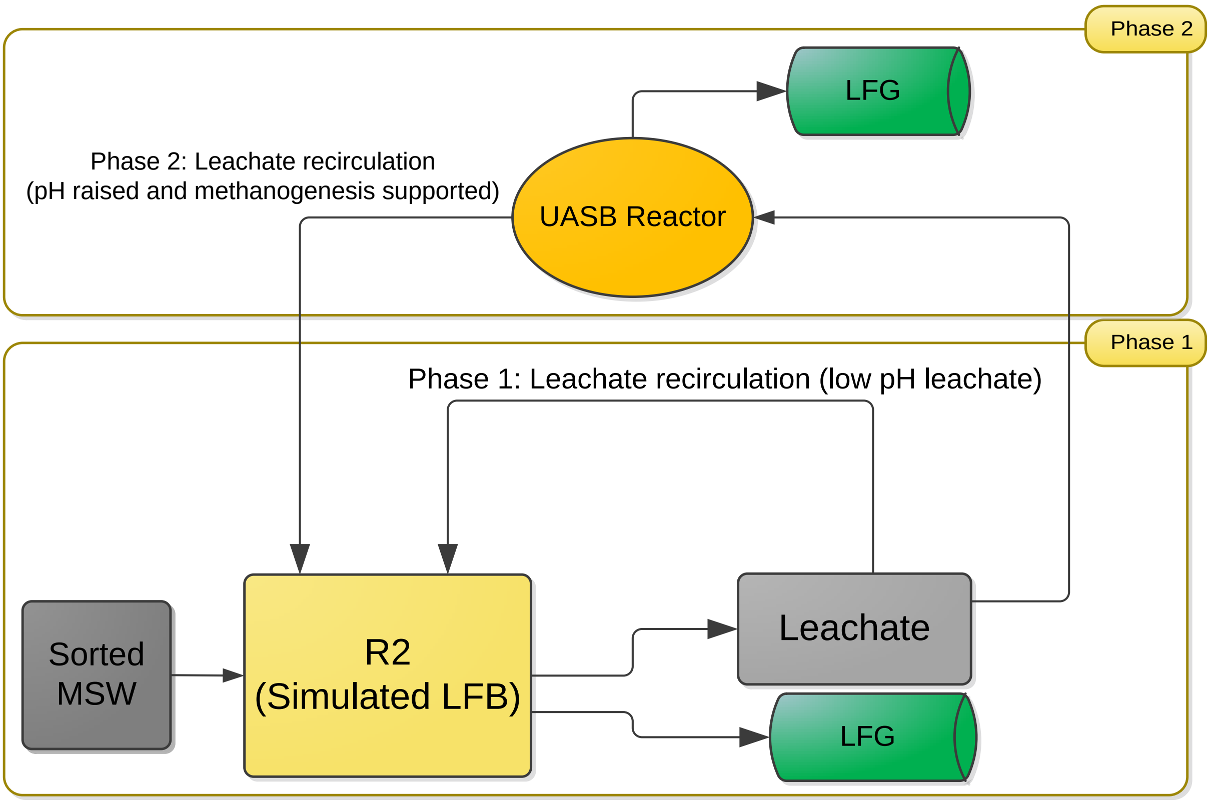


Figure 6-2: Flow diagram phase 1 and 2 of R2 (simulated LFB)

6.2.4 Analytical procedures

Temperature of the waste in the reactors was monitored weekly using Hanna Instruments Inc. 16.34 K-thermocouple thermometer with built-in microprocessor that has temperature probes specially designed to measure the temperature in compost. The probe was inserted into the reactor through the temperature/moisture content sampling ports and left for half a minute before readings were recorded. The ambient temperature was measured by a simple thermometer near and around the reactor. The temperature readings were all taken daily at 10:00 am throughout the operation time of the reactor.

pH of the leachate was determined daily for 13 weeks and thereafter weekly for 39 weeks making up a total of 52 weeks. pH was measured using SensIon pH meter model 156 Hach.

Leachate was analyzed for COD, Nitrogen-Ammonium and Phosphates. COD was analyzed using the dichromate method. Before analysis the samples were filtered with 4.4 µm folded paper filter (Schleicher & Schuell 595½). Phosphates and Nitrogen-Ammonium, were analyzed by spectrophotometry following the procedures as described in the Standard Methods (APHA 2005).

Samples for moisture content analysis of the incoming waste were taken during filling of the reactors and from the already filled waste, samples were taken after every four weeks for first three months of the study. In order to determine the moisture content the collected waste sample was weighed and oven dried at 103-105 ^oC after which the dried sample was weighed for quantification of evaporated water content.

The volume of gas generated was measured by displacement method. One inverted bottle filled with water was connected to the reactor via a 5 mm plastic pipe while another bottle received displaced water due to gas being bubbled into the inverted bottle. The volume of water displaced was measured and represented the volume of gas bubbled.

The gas composition (CH_4 and CO_2) was measured by using the inverted serum bottle liquid displacement technique. The displaced liquid was a strong 15% NaOH solution. A syringe was used to take a sample of biogas and injected into the serum bottle and as the biogas passed through the solution, the CO_2 was converted to carbonate and absorbed into the liquid. The CH_4 passes through the solution and an equivalent volume was pushed out of the top serum bottle. The displaced liquid was measured as the volume of CH_4 present in the biogas under the assumption that 1 ml CH_4 displaces 1 ml NaOH solution.

6.3 Results and discussion

This section provides the results and discussion of the monitored critical parameters COD, pH, temperature, nutrients, settlement of waste and the LFG generation in order to quantify the degree of waste stabilization in both reactors.

6.3.1 COD concentration variations in leachate produced from the reactors

The COD concentrations of the leachate collected at the bottom of the reactors as function of the time of stabilization are as shown in Figure 6-3. The COD concentrations of leachate in both R1 (control) and R2 (simulated LFB) were observed to increase from the initial 49,800 mg/l and 60,600 mg/l to 151,200 mg/l and 79,800 mg/l, respectively through the first 6 weeks.

The COD of R1 remained between 140,000 mg/l and 160,000 mg/l for 18 weeks in the 6th until the 24th week and began to fall to about 29,000 mg/l in the 33rd week and from there the COD was at an average of 30,000 mg/l. The COD concentration pattern of the leachate exhibited by R1 was somewhat different from what is theoretically expected of a sanitary landfill. In the aerobic phase in the beginning of the experiments the COD concentration is expected to rise but this aerobic phase only lasts a few days as the oxygen is depleted. Then, the waste becomes anaerobic and moves into the acidic phase and supports hydrolytic and fermentative reactions resulting in carboxylic acids and alcohols. During this phase the highest COD concentration is expected and as the acetogenic bacteria begin to convert these acids and alcohols to acetate, hydrogen, and carbon dioxide the COD is expected to begin to drop but the COD concentration of R1 took longer than expected to begin to drop as the phase was changing from aerobic to acidic and fermentation phase.

For the R2 reactor with recirculation, the COD concentration exhibited a pattern that conforms much more with the theoretical expectations. After the initial rise to 79,800 mg/l in the early phases of degradation, acids accumulated, pH dropped and remained low (as shown in later subsections of this section) and the COD concentration rose as high as 143,200 mg/l in the 16th week. At this point the reactor has become thoroughly acidified with a leachate pH so low that it inhibited methane production. A UASB reactor filled with anaerobic sludge was introduced to produce biogas from the already acidified leachate and to pre-treat the leachate before recirculation. From there on, the COD concentrations began to drop until levels as low as 8,500 mg/l in the 52nd week because of biogas production from the volatile fatty acids in the leachate. However, separate effects of the UASB on the COD concentrations of the recirculated liquid were not established because only the leachate collected at the bottom of R2 was monitored The findings from this study exhibited COD concentration patterns similar to the studies by San and Onay (2001) and Sponza and Ağdağ (2004) and Jiang et al.(2007).

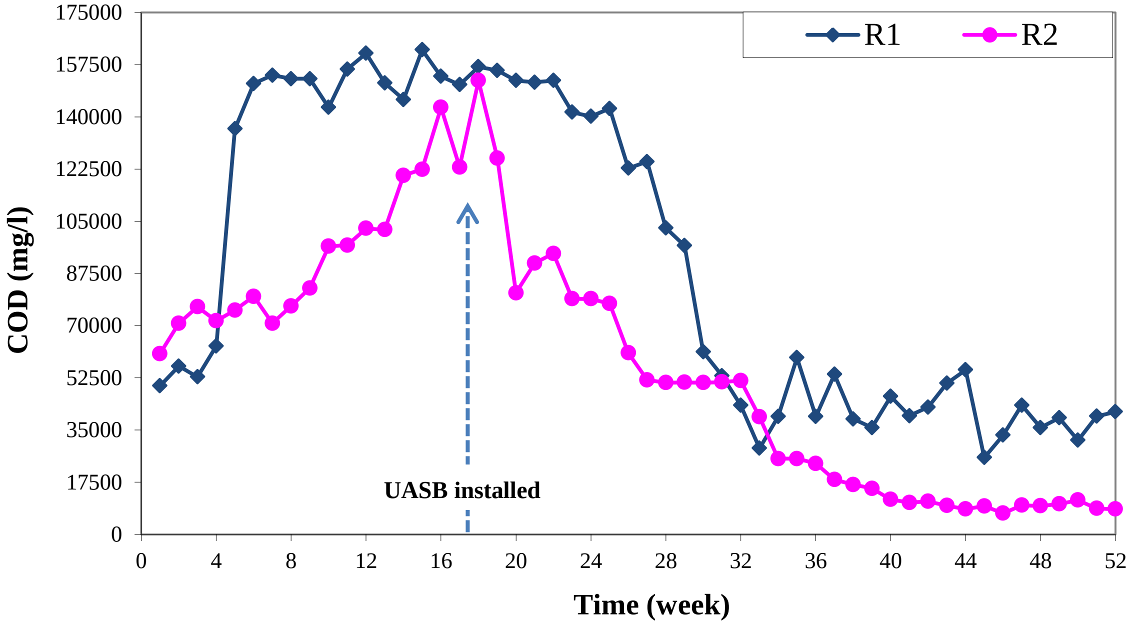


Figure 6-3: COD concentrations in leachate of R1 (control) and R2 (simulated LFB) as function of time

6.3.2 pH variation in leachate

The pH in the leachates of R1 and R2 as a function of time is presented in Figure 6.4. The initial pH in the leachate from R1 was 7.1. The pH value decreased for the first three weeks which was expected but then rose to 8.5 in the 10th week. The observation we have not been able to explain. Then a drop of pH to 5.2 by 15th week occurred probably due to imbalance of acidification and methanogenesis in the reactor. After the drop, the pH began to increase gradually and stabilized between 8.0 and 8.1 in the 46th week and onwards indicating utilization of VFA. The pH trend exhibited by R1 was typical for sanitary landfill leachate which was also observed by Jiang et al. (2007). An exception was the period between the 4th and 10th week with a temporary rather high pH value above 8.0.

The initial pH in the leachate from R2 was almost neutral at 7.2 as it was for R1. The pH values then decreased sharply to 5.5 until the 3rd week. The pH then maintained within a slightly acidic range of 5.2 and 5.6 for 18 weeks. pH in the acidic range is adverse to methanogenic activity. Such low pH values could be attributed to the production of low alkalinity, which is not enough for maintaining the neutral pH and buffering the VFA produced (Sponza and Ağdağ 2004). The low pH was presumably due to a constant dissolution and accumulation of VFA suggesting that acidogenic bacteria were governing the system (Veeken et al. 2000; Dinamarca et al. 2003; Valencia 2008). After introduction of the UASB reactor in week 17 the acids in the leachate were converted to methane in the UASB reactor and the resulting pH of the leachate began to increase to 7.3-7.7. The values were above neutral from the 32nd week onwards. Note that such pH conditions would then start favouring methanogenesis in the reactor now considered as a representative section of a cell would no longer be used for acidification purposes but rather for in-situ production of LFG.

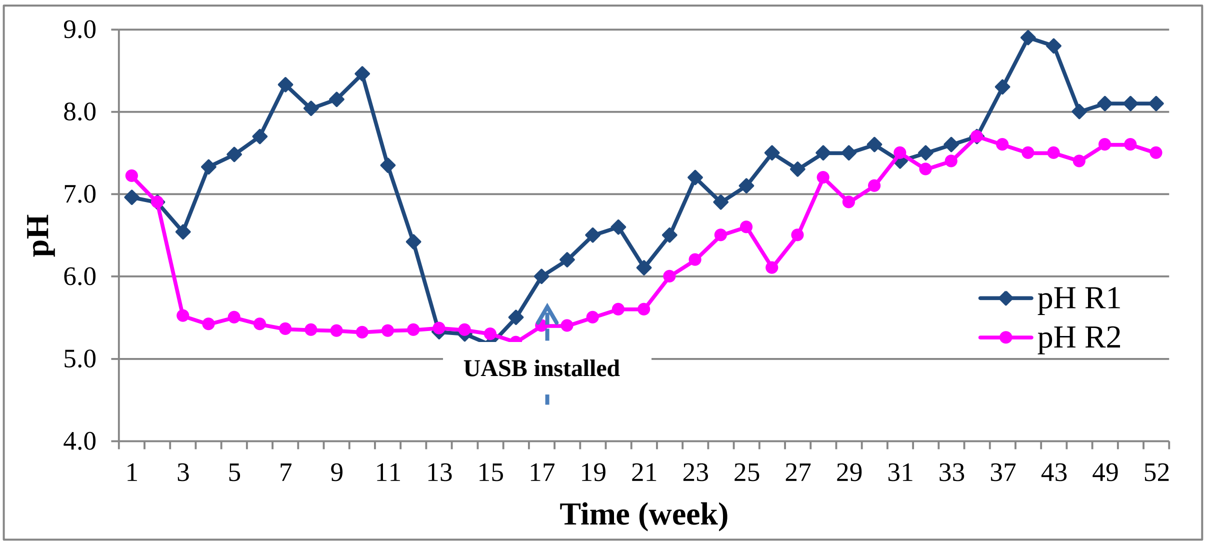


Figure 6-4: pH variations of the leachate in R1 and R2

Both R1 and R2 began with almost the same neutral pH and in the initial three weeks pH in both reactors dropped indicating the beginning of the acidic phase. As the degradation and low rate biogas production took place the pH of leachate from R1 began to rise while that of R2 maintained a low level in the acidic region and no gas was being produced. The pH of R1 then suddenly dropped into the acidic region to values the same as that of R2. The pH of R1 once again began to rise as was for R2 but the latter was due to the introduction of the UASB reactor coupled with recirculation of leachate. The pH in both reactors rose to above neutral and within the conditions favourable to methanogenesis.

6.3.3 Nutrients (N, P) variation in the leachate

The NH_3-N concentration found in the leachate of R1 and R2 as a function of time is given in Figure 6.6. The initial concentrations were found to be 324 mg/ and 432 mg/l for R1 and R2 respectively as a result of decomposition and leaching of organic nitrogen. The initial concentrations were more or less the same but due to heterogeneity of the conditions in the waste there was a difference in the concentration despite both reactors had been loaded simultaneously with almost the same mixture of waste. As a result of the decomposition, ammonia nitrogen concentrations in the leachate from R1 and R2 increased from the initial values of about 400 mg/l to a maximum of 538 and 1230 mg/l after 19 and 16 weeks of the study period respectively. The observed higher concentration in R2 is attributed to the recirculation of leachate which reintroduced ammonia back to the system and thus resulted in accumulation. Due to the fact that 100% of the collected leachate was recirculated, all the available nutrients in the leachate were contained and recirculated within the reactor.

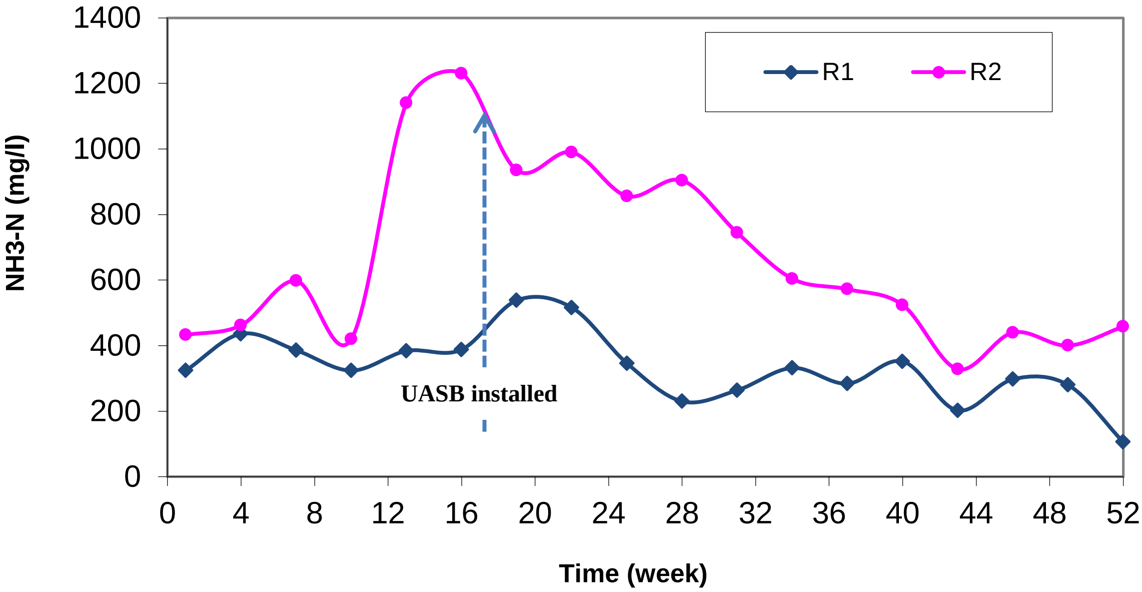


Figure 6-5: Ammonia-nitrogen variations in R1 and R2

After the 22nd week, the concentration of NH_3-N in the leachate in R1 began to decrease and reached 106 mg/l after the 52nd week due to possible depletion of nitrogenous matter in the waste. Similarly in R2, NH_3-N concentrations which were as high as 1,230 mg/l began to decrease in the 19th week that is after the introduction of the UASB reactor and remained at levels above 450 mg/l through to the 52 weeks of the study period. A gradual increase was expected but the NH_3-N concentrations dropped and continued to drop and we cannot explain the reason of the downward trend after the 18th week in R2 but a similar trend of decrease in concentration was observed by San and Onay (2001) and Sponza and Ağdağ (2004) whereby both studies had different recirculation rates and in some cases addition of water was done. It can be said that in landfills, the release of soluble nitrogen from solid waste into landfill leachate continues over a long period (Sponza and Ağdağ 2004). Leachate ammonia-nitrogen is a significant long term pollution problem that may cause inhibition of methanogenesis and may greatly determine when post-closure care of a landfill may be ended or reduced. This was also noted by Kjeldsen et al. (2002a) and Berge et al. (2007).

Phosphorus occurs in wastewaters almost exclusively as phosphates. Figure 6-6 shows the variations of the phosphate concentration in the leachate with time. The concentration of phosphates in the leachate of both reactors was more or less the same during the first 16 weeks. With the introduction of the UASB reactor to pre-treat leachate from R2 before recirculation, the phosphates concentration in the R2 leachate began to increase. This increase is mainly due to the recirculation of leachate which introduces back the phosphates that had already been released from the UASB reactor thus causing accumulation and further release of bound phosphorus. After the 16th week, while the concentration in R1 remained unchanged, the concentration in R2 increased which could be in part explained by the accumulation of phosphate due to recirculation. In the 40th week the concentration gradually began to drop as the pH was increasing to above 7.5 and phosphates were removed from the liquid phase by precipitation.

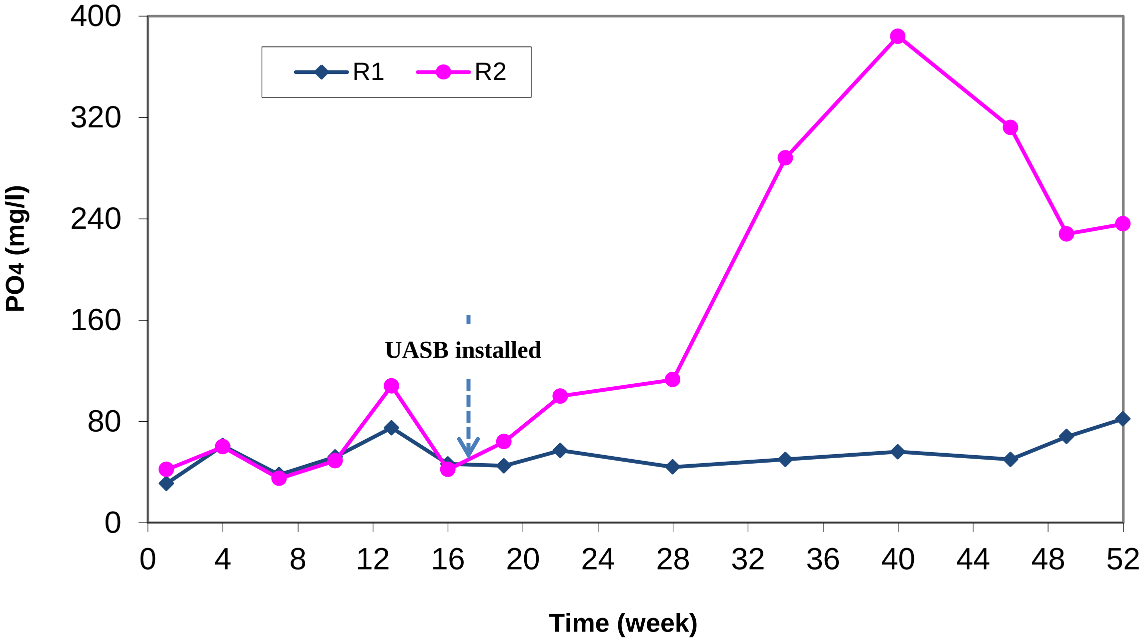


Figure 6-6: Phosphate variations in the leachate from R1 and R2

6.3.4 Temperature variation of the reactors

Variation of temperature of leachate from the reactors is shown in Figure 6-7. Both reactors exhibited a more or less a similar pattern. For both reactors R1 and R2, temperature initially increased to between 28 ^{o}C and 29 ^{o}C after 13 weeks of operation which indicated that the some heat was also being generated by the metabolism of microbes. As for R2, the temperature continued to rise and reached values between 36 ^{o}C and 37 ^{o}C (temperature optimum for methanogenesis) in the 34th to 40th week. The increase in temperature is probably due most importantly to the heat of the recirculated leachate. This heating is brought about by insolation of the exposed leachate storage and the UASB reactor during day time. However, during the 49th – 52nd week, the temperature in R2 was remarkably high despite the fact that the conversion was lower than during the previous weeks.

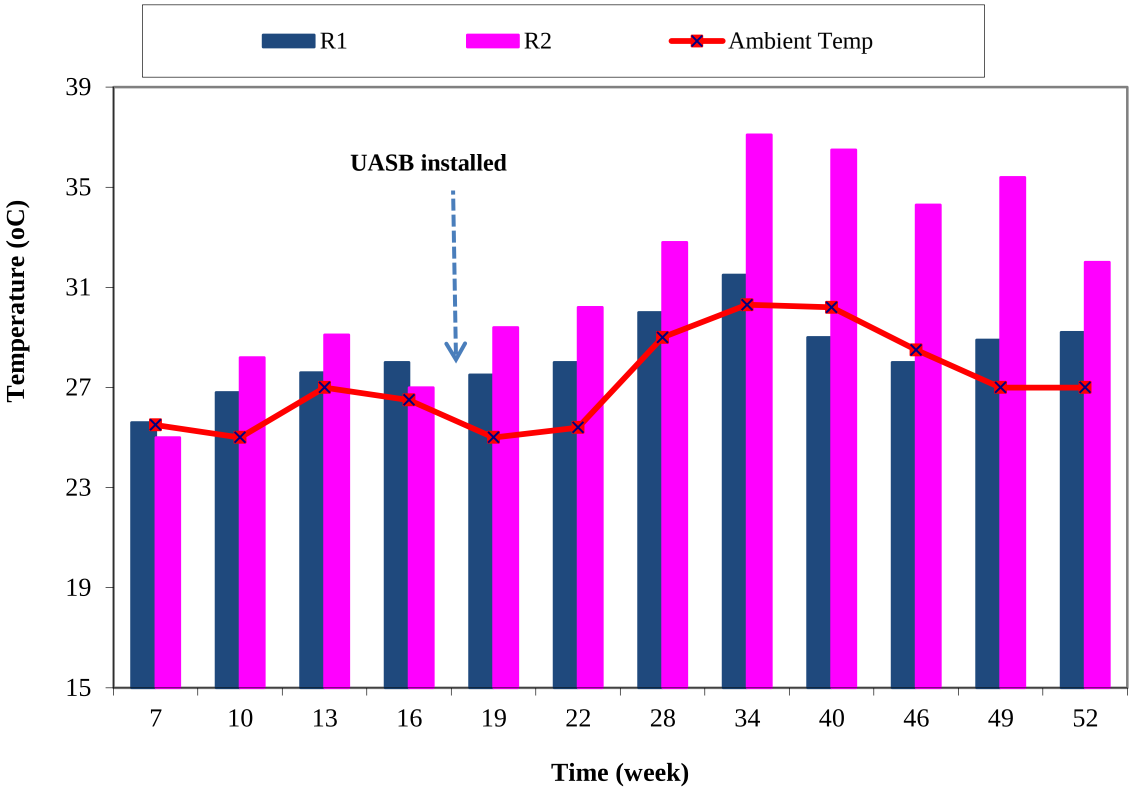


Figure 6-7: Temperature variation in the reactors and ambient temperature

6.3.5 LFG generation

The cumulative LFG generation of reactor R2 is presented in Figure 6.8. R1 began to generate LFG after 4-5 weeks while there was a lag in R2 which began to produce LFG after the 26th week of reactor operation. The generation of LFG in R1 was at a relatively low rate. The lag of LFG generation in R2 was mainly due to prolonged acidification of the reactor by recirculation of acidified leachate. This resulted in a pH below 6 (see Figure 6-4).

After the 16th week the introduction of the UASB reactor for ex-situ treatment of leachate from R2 brought about biogas generation via the UASB reactor. With the treatment of leachate, the pH of the leachate changed from acidic to near neutral. On the 26th week with the introduction of the UASB reactor biogas began to be generated at an average rate of 15 l/day as depicted in Figure 6-8. After 52 weeks of operation of the UASB reactor coupled to R2 produced cumulatively 12 m^3 of biogas and R1 about 0.15 m^3. It should be noted that, the volumes of gas generated from the UASB coupled to R2, may seem low because of problems of inadequate sealing of the reactors to avoid losses or leakages.

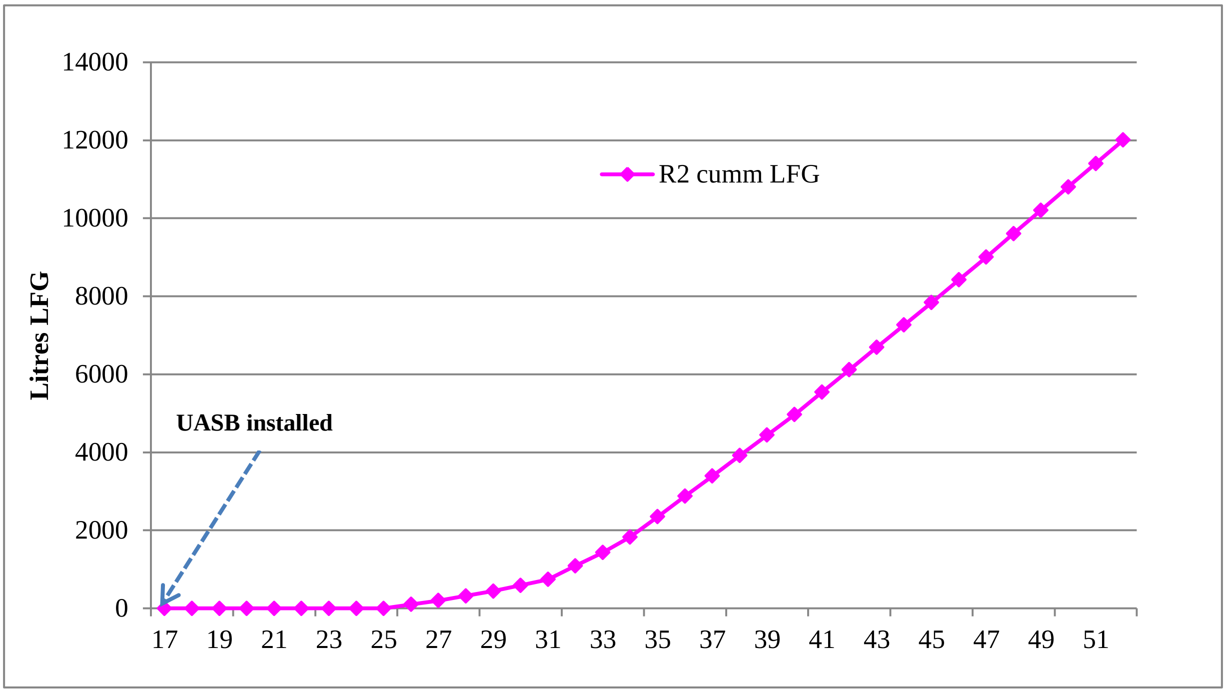


Figure 6-8: Cumulative LFG volume from UASB reactor coupled to R2 (simulated LFB**)**

Methane percentages for several samples are shown in Figure 6-9. The samples were taken after every 10 weeks of operation. The analysis shows that the typical composition of LFG was in the range of CH_4 35-46% for R1 and 48-55% v/v for R2, with an annual average of 42% and 51% v/v for R1 and R2 respectively.

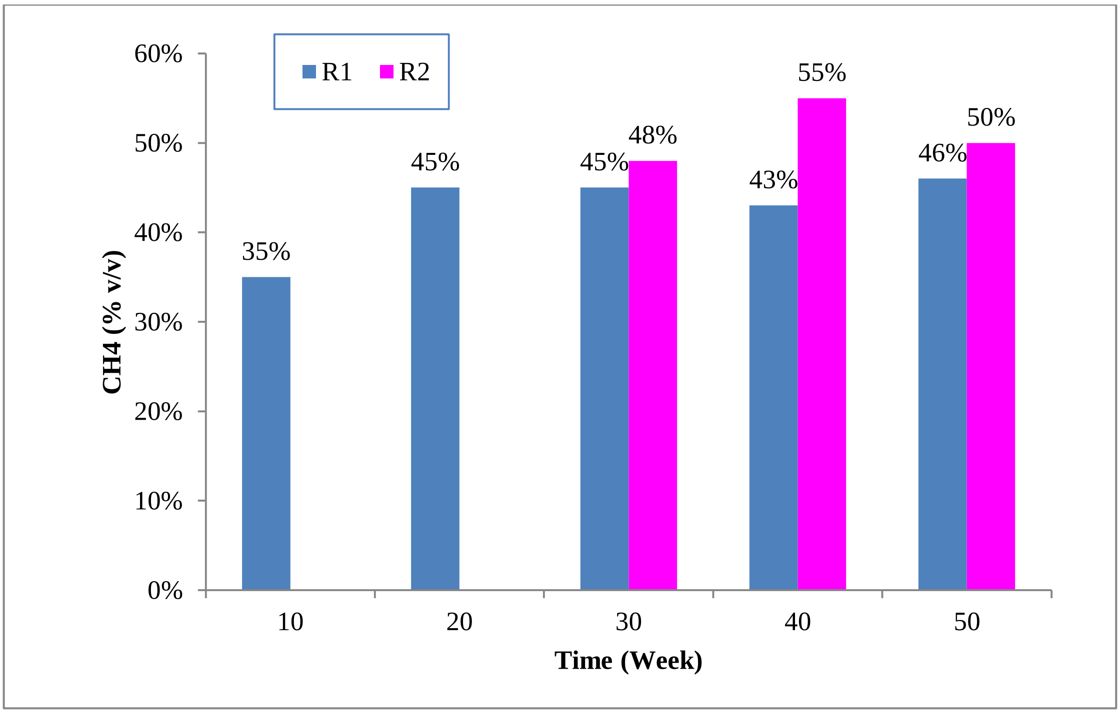


Figure 6-9: Methane percentages of several samples

6.3.6 Waste settlement and volume reduction

Settlement patterns observed for the reactors are illustrated in Figure 6-10. Initially, the settlement rate of the waste for the first three months in R1 and R2 was 0.75 cm/week and 1.59 cm/week respectively. Gradually the settlement dropped and no further settlement was observed after the ninth month of operation.

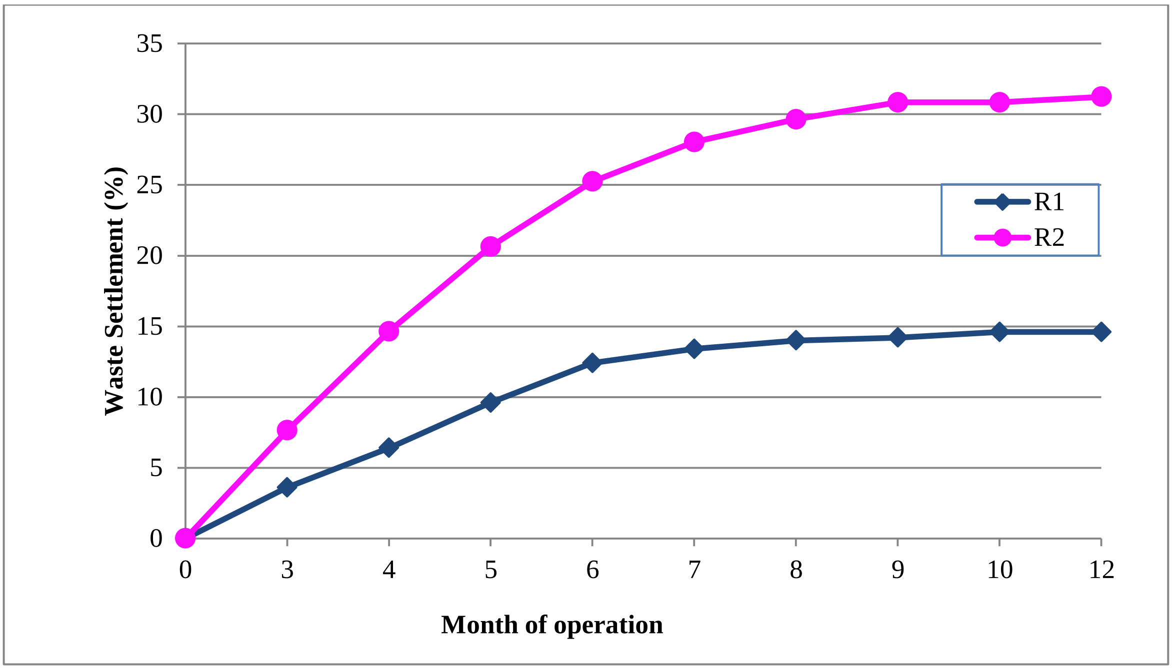


Figure 6-10: Settlement of the deposited waste with time

After one year of operation a total reduction in waste volume of 14.6 % in R1 and 31.2 % in R2 was observed. Waste settlement is a function of factors such as thickness and weight of cover and compaction, waste density and composition (particularly moisture), climate, etc. (Zhao et al. 2002). Settlement of waste in LFBs is a result of reduction in void space and compression of loose material due to overburden weight, volume changes due to biological and chemical reactions and dissolution of waste matter by leachate, movement of smaller particles into larger voids and settlement of underlying soils (McBean et al. 1995; Reinhart and Townsend 1998). In this study, availability of void spaces was limited by the compaction that was done during filling the reactors and due to leachate drainage. Settlement is also a result of the decrease of the remaining solids in the waste mass caused by degradation. COD reduction in the leachate is an indicator of the degradation of the waste taking place in the reactor. The settlement of waste provides an opportunity to utilize valuable air space prior to closure of the cell thus extending the life span of the entire landfill site.

6.3.7 Leachate generation

The leachate produced in the reactors as mentioned in section 6.2.1 was collected once a week from the bottom of each reactor and the volume was measured. Figure 6-11 presents the amount of leachate collected per week from each reactor during the entire period of study. The amount of leachate production per week from R1 reached a maximum in the second week operation and decreased afterwards as expected in a sanitary landfill where leachate production will be high in the beginning and the volume will gradually decrease until eventually no leachate was produced anymore. This point was reached after about 45 weeks. It may be assumed that at this point field capacity would have been reached. The cumulative amount of leachate collected from R1 throughout the study period added up to 702.15 litres from an initial amount of waste of 2394 kg wet waste. This figure means that the leachate production amounted to 293 l/ton of wet waste starting from an initial water content of 640 l/ton. Neglecting rainfall having entered the reactor and the mass reduction of waste due to degradation, the moisture content of the waste after about one year equals 640-293= 347 l/ton or 34.7%. This is within the 30 – 50% range of field capacity values presented in the literature.

In reactor 2 leachate production comprised the original leachate production that was also found in R1 and the recirculated leachate which may also influence the original amount of leachate produced. The latter amounted to a constant 91 l/week. The observed leachate production in R2 initially increased to about 110 l/week, then decreased and increased again to reach a level that varied between about 90 and 105 l/week after about 20 weeks. The decrease to a level of less than 80 litre/week during the period week 6 – 8 could not be explained but possibly clogging of the leachate collection system was the reason. Finally, after about 37 weeks a stable level of around 90 l/week was reached. This amount corresponded to the amount of recirculated leachate. From the leachate production figure R2 can be concluded that the recirculated leachate had reached the bottom of the reactor after about 2 weeks.

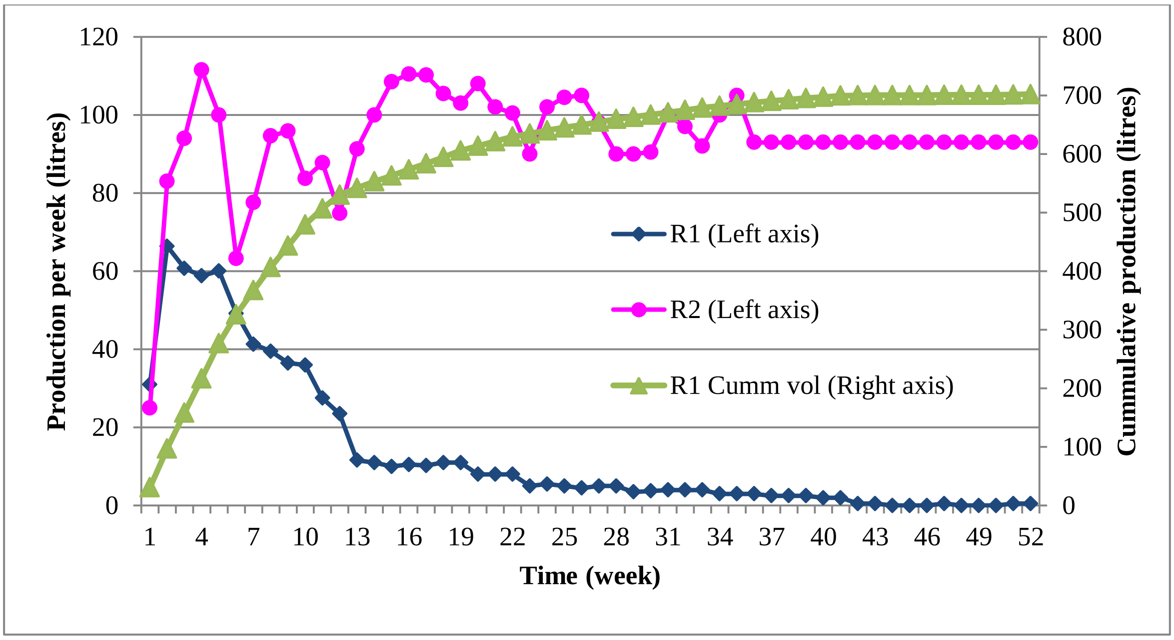


Figure 6-11: Leachate production in R1 (control) and R2 (simulated LFB) as function of time

6.4 Conclusions

The objective of this study was to identify the impact of leachate recirculation on waste degradation, methane production, and in situ leachate treatment, and to provide insights for the successful operation of LFBs in developing countries. The main results of this study indicate the validity and feasibility of operation of the LFB with waste characteristics of East

Africa to accelerate the stabilization of organic-rich wastes, enhance LFG production and achieve a degree of leachate treatment. Based upon results obtained during the study, the following specific conclusions are drawn:

1. The study confirms the literature with respect to the feasibility of the operation of a landfill as a controlled anaerobic bioreactor with leachate recirculation.
2. Leachate recirculation enhanced waste stabilization as reflected in higher gas production in R2 (simulated LFB) than in R1 (control) and more waste settlement.
3. Controlled acidification of the leachate is possible. The lesson learnt from the extended acidification of leachate in R2 and introduction of the UASB reactor can be taken as positive evidence that the LFB can be used for the first two steps of anaerobic digestion (i.e. hydrolysis and acidification) and the remaining step of methanogenesis can be carried out in a separate reactor.
4. In practice, this two stage approach of extended acidification means that no biogas is generated within the landfill so that there is no loss of methane from the landfill. Accordingly the two-stage process may result in a lower overall loss of biogas to the atmosphere.
5. Management of nutrients (N and P) requires attention because neither degradation nor removal of these parameters was observed in both R1 (control) reactor and R2 (simulated LFB).
6. The results obtained from this study come from a pilot-scale experiment. To confirm these results more experiments and probably a full-scale study are necessary to elucidate more precisely the LFB phenomenon.
7. The results obtained in this study are in general in agreement with results mentioned in literature for comparable experiments.

CHAPTER 7

LANDFILL BIOREACTOR: Identification, elaboration and comparison of innovative options

7.1 Introduction

Landfilling is currently the dominant disposal method of MSW in developing countries. Approximately 50% of the MSW generated in Tanzania is disposed in landfills (Salukele et al. 2011). Low costs and availability of land have made landfilling the most common waste management option in East Africa. Two main concerns associated with landfills are landfill gas (LFG) and leachate. Environmental and health issues related to leachate and landfill gas require reliable control measures. Advances in landfill research have indicated that the operation of landfills as bioreactors is a viable option for MSW management (Reinhart and Townsend 1998). The advances include leachate recirculation as a means of leachate treatment with an added advantage of faster waste stabilization and rapid LFG generation. Acceleration of waste decomposition leads to enhanced landfill stability and decreased landfill emissions coupled with regenerated and useable air space (ITRC 2005). The advantages of LFBs as compared to conventional landfills have been discussed in chapter 3.

This chapter presents four innovative concepts called system options of landfill bioreactor (LFB) systems making use of advanced existing knowledge and pilot-scale experimental results. The chapter also presents the schematic design and operation of these system options. The developed concepts comprise of materials recovery and transfer stations (MR-TF), large-scale LFB and other supporting systems for waste degradation, such as BIOCEL reactors, leachate storage and recirculation, leachate primary and final treatment and gas collection. The LFB and the BIOCEL process, leachate recirculation, various leachate treatment options and gas collection systems have been thoroughly reviewed in chapters 3 and 4. In chapter 5 the models to calculate biogas production in landfills have been elaborated. The development of these innovative system options has been achieved through the following: a) empirical research conducted in East Africa - Tanzania to diagnose the existing MSW management practices and amount and characteristics of collected MSW (chapter 2); b) extensive literature review on LFBs and leachate treatment (chapters 3 and 4); c) gas production modeling (chapter 5) and d) a locally conducted pilot-scale study (chapter 6).

Four different system options are proposed. They have aspects in common and also several distinctions. The commonalities are:

- Type of waste and pre-treatment in terms of sorting and segregation;
- Landfill bioreactors filled in cells and lifts with leachate recirculation at a low flow-rate using vertical recirculation wells ;
- Usage of vertical gas extraction pipes.

The systems are different as to the location of LFG generation as explained below:

- System option 1: Standard landfill bioreactor (section 7.2);
- System option 2: Standard LFB with part of LFG production in a UASB reactor at the LFB site (section 7.3);
- System option 3: Two-stage treatment – Centralized BIOCEL followed by a LFB (section 7.4);

- System option 4: Two-stage treatment – Standard LFB fed by decentralized BIOCEL reactors at transfer stations (section 7.5).

The standard LFB of System option 1 is geared towards having MSW treatment and enhanced biogas production with significant reduction of GHG emissions. System option 2 adds to System option 1 the generation of biogas from the acidified recirculated leachate in a separate reactor (UASB) and up to 100% collection of the biogas. In this way the loss of gas that occurs from the large surface area of the LFB can be significantly reduced. System option 3 incorporates a centralized BIOCEL reactor (see chapter 3). The BIOCEL-system assists in the rapid biodegradation of the easily biodegradable organics and thus reduces the tonnage of waste to be landfilled in the LFB. The BIOCEL-system also reduces the possible loss of gas generated at the LFB. The setup of System option 4 is more or less similar to System option 3 with a difference on the size and number of BIOCEL reactors. In this system, decentralized BIOCEL-systems are coupled to the transfer stations whereby the waste is pre-treated before being transported for final disposal in the LFB.

Prior to being fed to the systems described above, the collected MSW is taken to a material recycling and transfer station (MR-TF). The sizing of the MR-TF can be based on the population to be served or the annual tonnage of waste to be handled. Based on population, the envisaged MR-TFs are aimed at serving in areas with at least 100,000 people within a locality and spatially distributed covering all the city's MSW collection areas. Alternatively, the MR-TF should receive at least 100,000 tons wet waste per year. Thus, a city like Dar es Salaam, Tanzania, would have 10 of these MR-TFs based on the current MSW generation rate of an estimated 3,000 tons/day. The alternative sizing of the MR-TF is used in the proposed system options in this thesis. Under Tanzanian conditions reusable and recyclable materials recoverable at MR-TFs make up to about 6% of all collected waste (see chapter 2 of this thesis). They are:

- Plastics – soft polymer plastics material in bag or liner form (e.g. PE bags and sheets including LDPE and HDPE), PET bottles and containers, LDPE and HDPE bottles, containers and caps and PVC based plastic materials;
- Textiles – pieces of clothes, rugs, pieces from tailoring marts (rags);
- Metals – ferrous, non-ferrous metals and aluminium cans;
- Glass – glass bottles (clear, green, brown) and jars for food and beverages.

The separated plastics, metals and glass have to be directed to the respective recycling industries while the textiles are to be reused to make other fabrics by local tailoring marts.

Table 7-1 presents general input data for enabling the calculations for comparison and evaluation of the options proposed in this thesis. The conditions of Dar es Salaam have been taken as point of departure. The required information for calculations are derived from the empirical research (chapter 2), from literature compiled in chapters 3 until 5 and the field experiment described in chapter 6. The composition of the waste discussed in this section and in Table 7-1 is drawn from the percentage distribution chart presented as Figure 2-4 in chapter 2 of this thesis.

The waste is assumed to have a density of 0.5 ton/m^3 and a mass flow of 1,000,000 tons/year. The amount of waste emanating from the MR-TFs after removal of valuable products (60,000 tons) is about 940,000 tons wet waste per year. Out of the 940,000 tons and based on percentages presented in Table 7-1 the waste to be treated consists of 234,000 tons dry biodegradable organic matter, 640,000 tons water (64% moisture content - chapter 2 and 6) and 66,000 tons inert materials (organic and inorganic).

Table 7-1: General input data based on the situation of Dar es Salaam

Description	Value
Amount of collected MSW	1,000,000 tons/year
Amount for landfilling (from MR-TF)	940,000 tons/year
Composition on dry basis*	65% Biodegradable organic 35 % Inert (non-biodegradable) • 29% Inert organic • 6% Inert inorganic
Density of waste	0.5 ton/m^3
Composition on wet basis (1 ton)**	640 kg water 360 kg dry matter
Distribution of the dry matter	234 kg (i.e. 65% Biodegradable organic) 104.4 kg (i.e. 29% Inert organic) 21.6 kg (i.e. 6% Inert inorganic)
At MR-TF 6%*** of the waste is removed as (dry) inert waste (60 kg)	53 kg inert organic (textiles and plastics) 7 kg inert inorganic (metals and glass)
Waste composition remaining after MR-TF in 940 kg that goes to the LFB or BIOCEL	640 kg Water 234 kg (Biodegradable organic) 51.4 kg (Inert organic) 14.6 kg (Inert inorganic)
Waste composition per 1 ton of waste	681 kg Water 249 kg (Biodegradable organic) 55 kg (Inert organic) 16 kg (Inert inorganic)
Biodegradable organic fraction (1 ton)	
Slowly	25% (62 kg/ton)
Moderately	42% (105 kg/ton)
Rapidly	33% (82 kg/ton)

*The composition on dry basis is established from chapter 2, Figure 2-4 assuming that only 50% of the grass/leaves are biodegradable organics.
** Moisture content of the collected waste was found to be 64% (chapter 2)
*** Recyclables and reusable removed from the waste stream (chapter 2)

The inert organics notably include paper waste and 50% of grass and leaves (50% of the grass and leaves are assumed to be biodegradable organics). Paper could have been included in the recyclables but in Tanzania, recycling of waste paper cannot be beneficial due to long distances of haulage of the recovered paper to the pulp and paper industries. Tanzania has two such industries, one in Moshi district-Kilimanjaro region and another in Mufindi district-Iringa region at approximately 530-590 km from Dar es Salaam city.

The dry biodegradable organic matter contains fractions with biodegradation rates (i.e. slowly, moderately and rapidly) ranging from 0.01 to 0.7 yr^{-1}. The subdivision into three biodegradable fractions given in Table 7-2 was adopted with some modifications from Table 5-2 (chapter 5) and Table 2 in Scharff et. al (2006). The OM content given by Scharff et. al (2006) is based on the total amount of wet waste (from Europe), whereas for this thesis we took 249 kg biodegradable organic matter/ton of wet waste corresponding with 234,000 tons of dry biodegradable OM per year as point of departure. The degradation rate constants and the conversion factors are adopted from literature sources discussed in chapter 5.

Table 7-2: Degradation rate constants and conversion factors of the three biodegradable fractions and organic matter content of MSW adopted for the multi-phase model

Type of organics	**Degradation rate (k)** (year^{-1})	**OM content (X_o)** (kg OM/ton wet waste)	**Conversion factor (f)** (m^3 LFG/ kg OM)
Slowly	0.02 (k_s)	62 ($X_{o,s}$)	0.70 (f_s)
Moderately	0.2 (k_m)	105 ($X_{o,m}$)	0.72 (f_m)
Rapidly	0.4 (k_r)	82 ($X_{o,r}$)	0.74 (f_r)

The potential biogas production per ton of waste for the three biodegradable fractions is given by the amount of biodegradable organic matter per ton of wet waste (obtained after removal of 6% inert waste) calculated in Table 7-1 and the conversion factor in Table 7-2.

Table 7-3: Potential biogas per ton of deposited waste

Biodegradable OM fraction	**Conversion factor (f)** (m^3 LFG/ kg OM)	**Potential biogas production** (m^3 LFG/ ton wet waste)
Slowly	0.7	43.4
Moderately	0.72	75.6
Rapidly	0.74	60.7
	Total	**179.7**

The potential biogas production for each biodegradable OM fraction is given by the product of conversion factor (f) and OM content (X_o) as shown in Table 7-3. The sum of the produced potential biogas is 168.48 m^3 LFG/ton wet waste in 1,000,000 tons original wet waste/year that goes to the MR-TF. This is equivalent to 179.7m^3 LFG/ton wet waste in the 940,000 ton wet waste per year that is transported to the landfill at an average conversion factor of 0.72 m^3 LFG/ kg OM.

In order to enable comparison for the four system options the following aspects are considered:

- Conversion of dry organic matter;
- Production of biogas (LFG);
- Collection of biogas;
- Contribution to greenhouse gas emissions (GHG);
- Amount of leachate;
- Leachate treatment;
- Qualitative costs analysis;
- Size of the landfill site;
- Size of the BIOCEL reactors.

7.2 System option 1: STANDARD LFB

7.2.1 General layout of option 1

Option 1 is an advanced and modernized landfill whereby the input is sorted waste from Materials Recovery and Transfer Stations (MR-TF) hauled by designated trucks to the landfill site. The waste is then landfilled and left to decompose under controlled conditions. Leachate is collected and recirculated back into the landfill site. This type of landfill can be considered as a bioreactor which maintains the appropriate moisture content to stimulate microbiological activities and simultaneously treats leachate in-situ. LFG generated within the bioreactor is collected at the top and is cleaned before being put to valuable use. Schematically (Figure 7-1), System option 1 is comprised of a MR-TF, a centralized LFB, leachate storage tank for equalization of the leachate recirculation rate, LFG collection system and a leachate treatment plant (LTP). These components are discussed in the subsequent subsections.

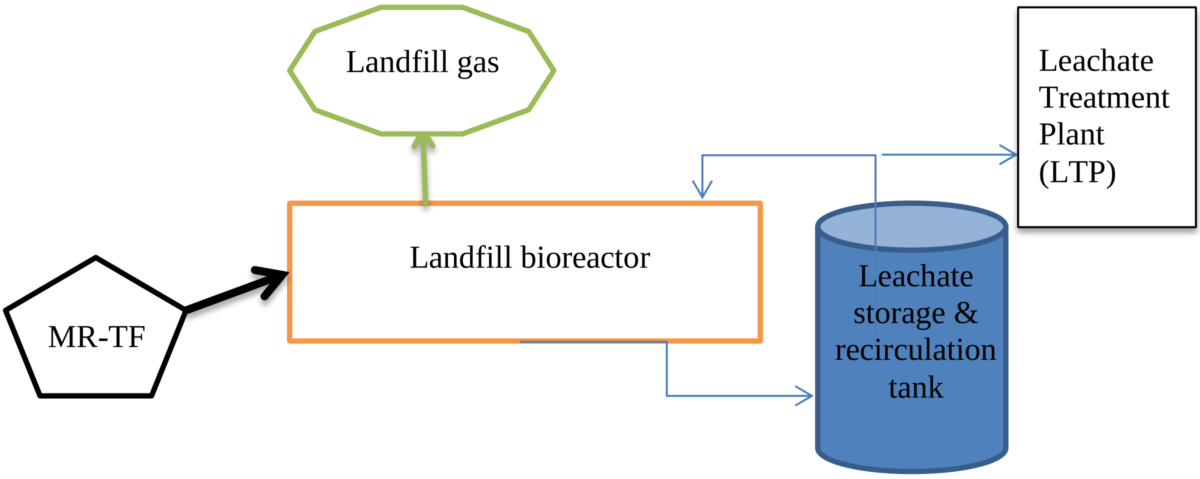


Figure 7-1: System option 1 – Standard Landfill Bioreactor

7.2.2 Materials Recovery and Transfer stations Facility

MSW collected from waste generation points (residential and commercial premises) is transported to the transfer stations as is the current practice in many large cities of developing countries. Dar es Salaam, Tanzania, is no exception. At the transfer station, recovery of materials is established where a part of the non-biodegradable materials are sorted and removed from the waste input stream of the LFB. The functions of these MR-TFs depend directly on the materials to be separated and recycled. It is important therefore to develop recycling opportunities, so that non-biodegradable and valuable materials are deviated from landfills as much as possible. In following calculations it has been assumed that the amount of waste envisaged to be taken out of the collected waste stream in the MR-TFs amounts to 6% so that on a total stream of 1 million tons per year 60,000 tons is separated for recycling and 940,000 tons/year is sent to the landfills. The latter is mainly composed of different fractions of OM and water.

The design of MR-TFs is based on the combination of manual and mechanical separation methods. Currently, in the Tanzania and East Africa, mechanical separation is not a preferred option mainly due to power/energy reliance and proneness to mechanical failures coupled to lack of resources for operation and maintenance.

The MR-TF flow diagram proposed in this chapter (Figure 7-2) refers to handling of both packed and unpacked commingled and source-separated (SS) (present and future) MSW. At the receiving area waste in bags and sacks is opened up and all wastes are subsequently forwarded to the sorting workshop. Here, the wastes are separated in a recyclable and a non-recyclable fraction. The latter consists of mostly biodegradable material (food and non-food biowastes). This goes to the landfill and the former to recycling industries.

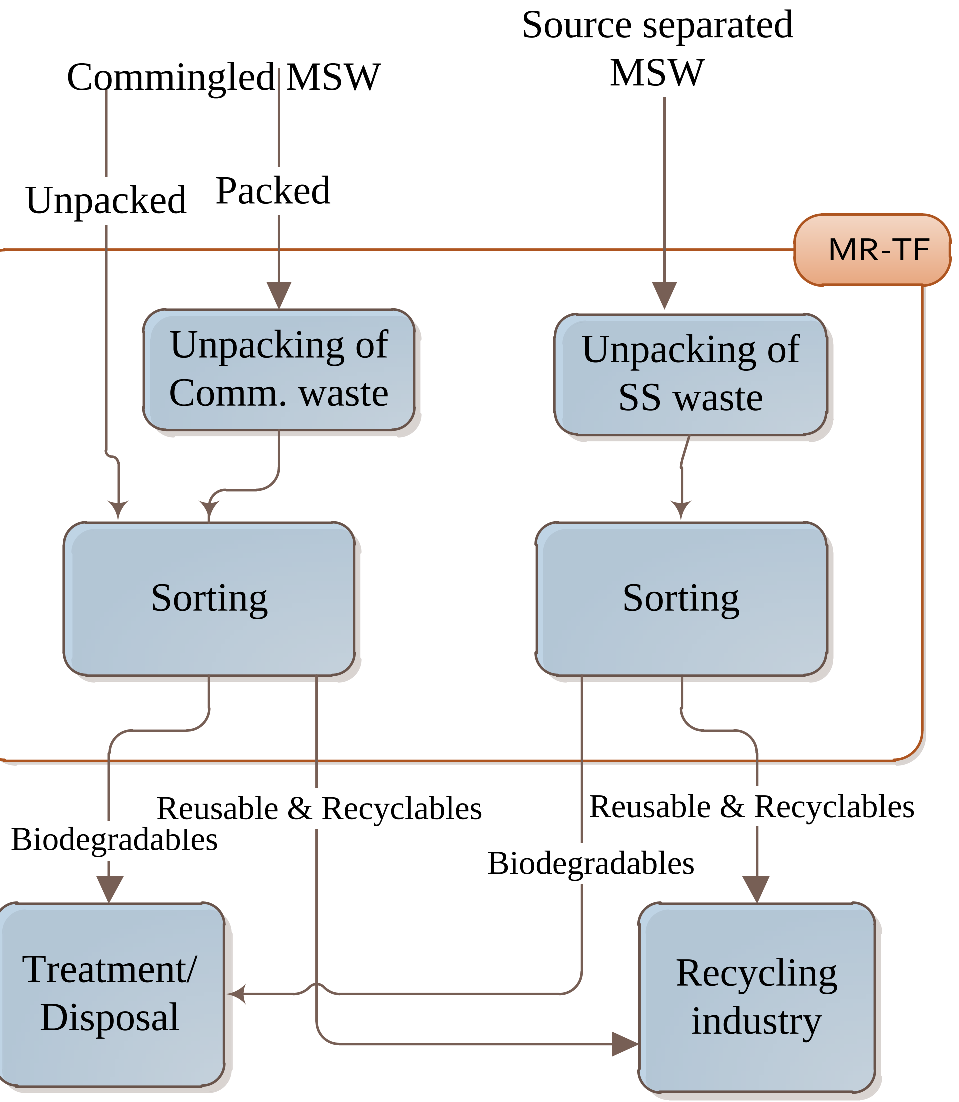


Figure 7-2: Process flow diagram of MR-TF

7.2.3 Landfill Bioreactor (LFB)

The LFB is an accumulating batch-reactor system consisting of sections made up of cells provided with gas extraction and leachate collection and recirculation pipes. The cells are gradually filled with sorted MSW received from the MR-TF as conventional sanitary landfill operation guidelines prescribe and capped after filling. The recommended height of cells is 10 m excluding intermediate and final cover. Active cells are provided with gas extraction and leachate collection and recirculation systems. The leachate recirculation and gas collection is started once a cell is filled and capped. The design and set up of these systems, the specification in terms of the sizes of pipes, spacing in a given surface area and other appurtenances such as pumps and valves are discussed in chapter 3.

It is assumed that one cell per week is prepared to receive fresh waste from the MR-TF. Taking a total deposition of 940,000 tons MSW/year, each week a cell is filled with 18,000 tons. The surface area required to fill the 18,000 tons of waste with a density of approximately 0.5 ton/m^3 and 10 m high is 3,600 m^2/week equivalent to a land requirement of 18.8 ha/year.

For the sake of the calculations it has been assumed that we start with the disposal at an empty site. Each week a new cell is installed, filled with waste, capped and taken into operation. In this way the cell becomes active, which means the cell is producing biogas. After N cells have been filled and taken into operation, installing a new cell is combined with uncoupling the oldest cell from the total system. In this way a stationary situation is obtained with N cells always in operation as depicted in Figure 7-3. For this stationary situation we can calculate the biodegradation of organics and production of biogas. The life time of an active cell is $N/_{52}$ years. Considering the full life of a landfill bioreactor there are four types of cells:

- New cells receiving fresh waste from MR-TF and capped after one week;
- Fully active cells where leachate is collected and recirculated and LFG is collected;
- Partially active cells where leachate is not recirculated but LFG is collected;
- Closed cells without collection and recirculation of leachate and without LFG collection.

It is envisaged that a fully active cell is in operation for 5 years. After that leachate recirculation is stopped so that the cell becomes partially active. The cells becomes partially active because LFG is still generated and collection continues for another 5 years. Accordingly, it takes 10 years before the cell is totally closed. It should be noted that the closed cells shall continue producing gas but this gas will not be collected as it is a minor flow. Figure 7-3 is a description of the operation of the landfill site on a weekly basis whereby each block represents a cell filled per week and the assumption is that the cells are active (full and partially) for a period of 10 years. This operation period of 10 years corresponds with 10 times 52 active cells equaling 520 cells. The description of Figure 7-3 is summarized as follows:

1. Time is 1 year: all 52 cells are active and 1 new cell is receiving fresh waste from the MR-TF;
2. Time is 5 years + 1 week: first cell is partially active, 260 cells (5 years) are fully active and 1 new cell is receiving fresh waste;
3. Time is 6 years: first 52 cells (1 year) are partially active, 260 remaining cells are active and 1 new cell is receiving fresh waste;
4. Time is 10 years + 1 week: first cell is completely closed. 260 cells (5 years) are partially active, next 260 cells (5 years) are active and 1 new cell is receiving fresh waste and after 10 years the waste in the cell is expected to be at field capacity;
5. Time is 11 years: 52 first cells are completely closed, 260 cells (5 years) are partially active, 260 next cells (5 years) are fully active and 1 new cell is receiving fresh waste;
6. Time is 15 years: 260 cells (first 5 years) are completely closed, next 260 cells (5 years) are partially active, the following 260 cells (5 years) are fully active and 1 new cell is open receiving fresh waste.

After 10 years we have achieved a more or less a stationary situation. That means from that moment on we always have 260 active and 260 partially active cells. The closed cells however will still produce biogas and contribute to the emission of GHG to the atmosphere but at a significantly reduced rate.

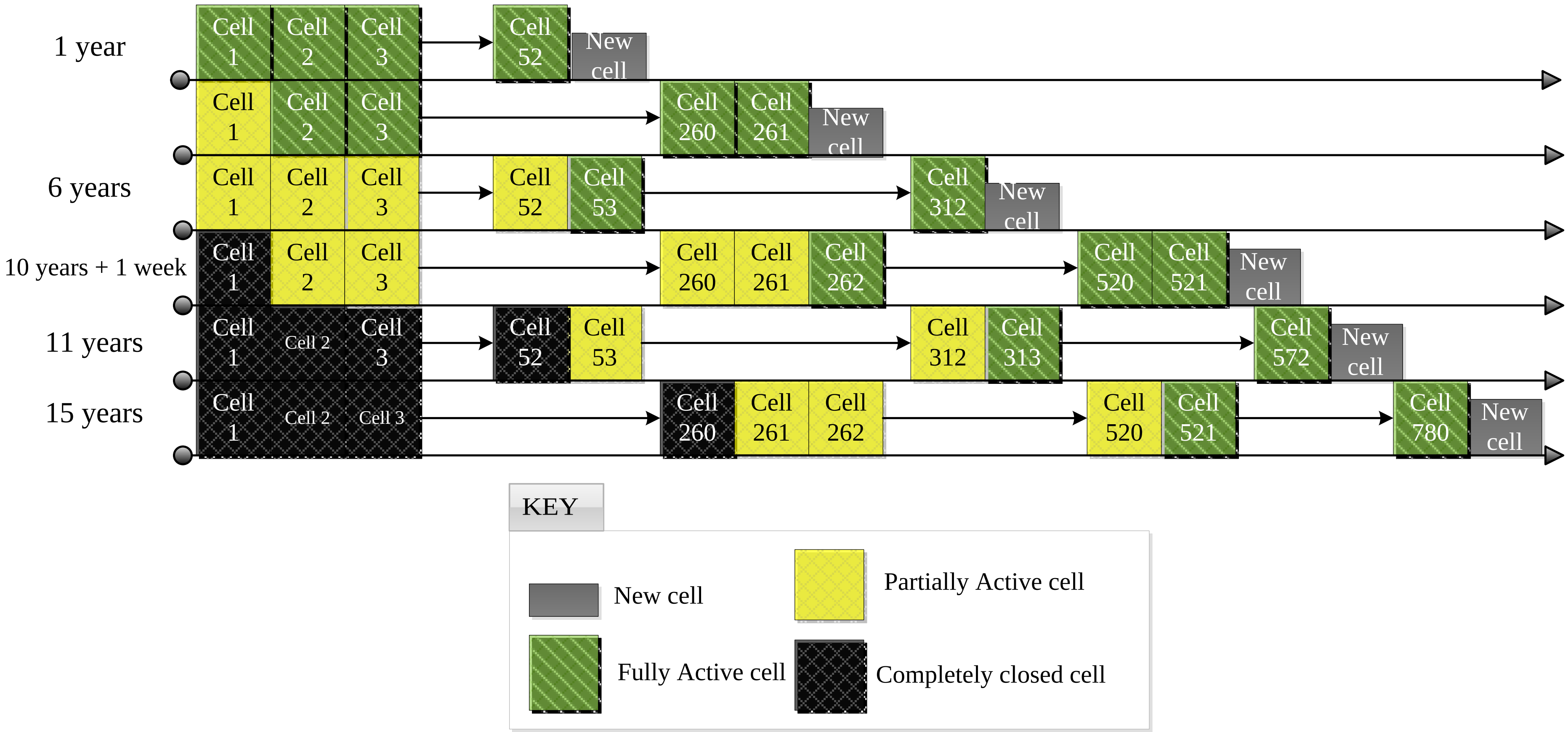


Figure 7-3: Mode of opening active cells and closure of in-active cells for a 10 years operational period

Waste conversion in the LFB

The processes taking place in the LFB lead to a biodegradation of the organic matter and will affect the moisture content. The conversion processes result in the production of LFG and leachate. The inert organics and inorganics originally present in the waste collected will not be affected. The biodegradation of organic matter with time is approximated here using a first order reaction. The rate of degradation depends on a host of factors among which the temperature is prominent (chapter 3). For the field conditions prevailing in tropical countries close to the equator, degradation constants of 0.02, 0.2 and 0.4 yr^{-1} for slowly, moderately and rapidly biodegradable OM respectively are assumed (chapter 5 section 5.3.2, Table 7-2).

During stabilization of the waste also the water content of the waste is undergoing important changes. During filling of the individual landfill cells water is squeezed out of the oversaturated waste due to the pressure increase of piling up wastes in the landfill. This water will leave the landfill as leachate. Some water is released as it takes part in the degradation of organic matter. In addition, negligible amounts of water will leave the landfill as vapour during the formation of landfill gas. As shown in chapter 4 the latter two amounts of water are small under Tanzanian conditions in comparison with the water squeezed out as leachate and are therefore neglected.

7.2.4 LFG generation in System option 1

In this paragraph we discuss the biogas production potential, the expected production and the greenhouse gas emissions. The calculations are followed by a sensitivity analysis.

LFG potential

The LFG production potential is based on the annual disposed amount of biodegradable organic matter (dry weight) of 234,000 tons/year . The average conversion factor of organic matter in the waste is 0.72 m^3 LFG/kg OM converted (Table 7-3). Therefore the LFG potential is 234,000*0.72*1000 = 168,480,000 m^3 LFG/year at STP (0^o C and 1 atm.) i.e. assuming all the waste has the same conversion factor. For the calculation of the global warming factor (GWF) it is important to calculate the percentage of LFG that is finally recovered and the percentage of LFG that is not recovered and that is emitted to the atmosphere.

LFG production

For calculation of the annual amount of LFG produced from the active cells, the following procedure is followed.

- Each year 52 new cells are filled with 940,000 tons of waste. This amount of waste may potentially produce 168,480,000 m^3 LFG/year;
- Each year 52 partially active cells with an age of 10 years are closed;
- The total lifetime of the active cells is 10 years.

The LFG amount (m^3/year) produced from the closed cells with an age of N years can be calculated using the multi-phase equation (5-12) derived in chapter 5, which is given by:

$$Q = M_o * \left(X_{o,s} * f_s * (1 - e^{-k_s * t}) + X_{o,m} * f_m * (1 - e^{-k_m * t}) + X_{o,r} * f_r * (1 - e^{-k_r * t})\right)$$

Originally, the amount of waste deposited M_o is 940,000 tons/year. In the equation $X_{o,s}$ is the slowly biodegradable organic fraction, $X_{o,m}$ is the moderately biodegradable organic fraction l, and $X_{o,r}$ is the rapidly biodegradable organic fraction whose values are obtained from Table 7-2 and t = 10 years. We look also at the potential amount of gas that can and will escape into the atmosphere. In spite of 50% oxidation by the top cover it is envisaged that a fraction of the gas will escape to the atmosphere.

The LFG production is evaluated on the basis of 10 years of operation. The LFG production at STP (during the active period of the cells) obtained from the complete site can be calculated using Equation 5-12. The LFG production amounts to 124,836,084 m^3 LFG/year which is 74.1% of the potential LFG. From this amount, 80% is assumed to be collected and 20% escapes. Out of the non-collected part 50% is converted into CO_2 and 50% emitted to the atmosphere as CH_4.

We consider the situation that the active operation time of the disposed waste is 10 years with 124,836,084 m^3 produced LFG per year (74.1% of the potential amount). Practically, the LFG collection efficiency is 80%. Therefore the actual amount of collected LFG is 99,868,867 m^3 about 59.3% of the potential amount of LFG present in the waste. The remaining unconverted and uncollected LFG after an active operational time of 10 years that will eventually be partly emitted as methane and carbon dioxide is 68,611,133 m^3. These values are shown in Figure 7-4.

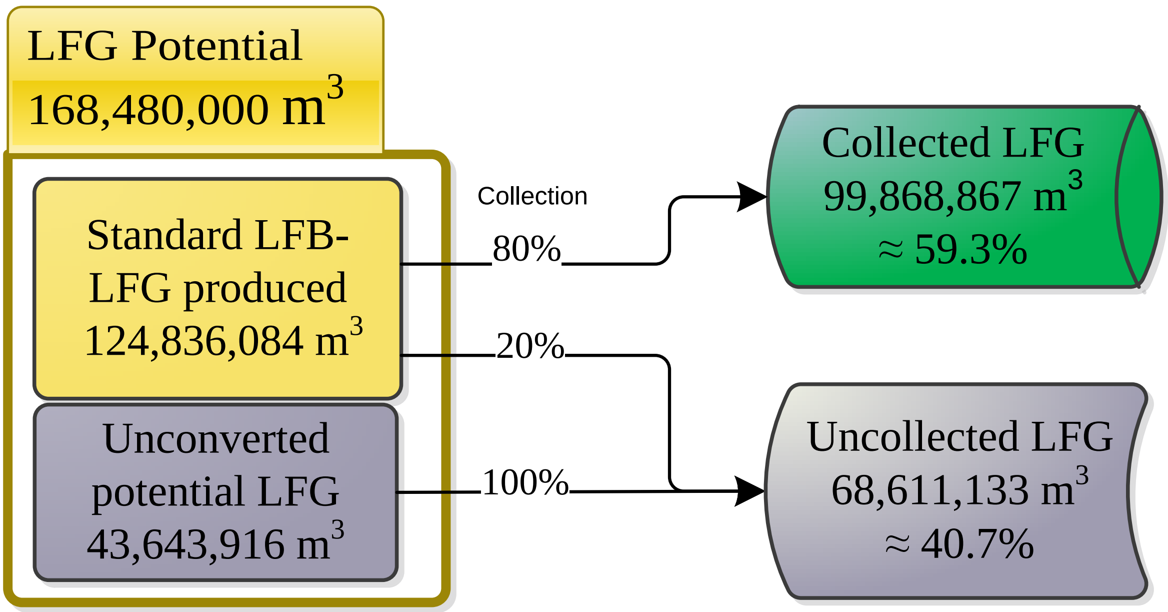


Figure 7-4: System option 1 – Volumes of collected and uncollected LFG during a 10 years operational period.

Estimation of Greenhouse Gas (GHG) emission associated with Standard LFB

Methane emission from landfills is one the major contributors to the Greenhouse effect in the world. In the context of this thesis, methane from the LFB is considered to be a GHG, but carbon dioxide is not. Carbon dioxide emission from the waste mass is a biogenic by-product and as such not included as part of the Global warming factor (GWF). It is assumed that emission of biogenic CO_2 is neutral to global warming, because the CO_2 originates from organic matter generated by an equivalent biological uptake during plant growth (IPCC 2006). When organic materials derived from biomass sources are landfilled, a portion of the carbon compounds in these materials does not decompose. Under natural conditions, virtually all of the material would decompose aerobically, and the carbon would be released as biogenic carbon dioxide. When the materials are landfilled, aerobic biodegradation is prevented. The carbon in those materials that does not fully decompose (anaerobically) is removed from the global carbon cycle, and is said to be stored. It is counted as an anthropogenic sink equivalent to removing CO_2 from the atmosphere.

The GWF from LFG emissions for this system option is estimated by considering the methane emission and its Global warming potential (GWP). GWP is a measure of the contribution to global warming of a certain mass of a greenhouse gas (GHG) (in this case of methane) as compared to that of an equivalent mass of carbon dioxide (kg CO_2-eq. per ton disposed waste). Methane emissions include: fractions of methane uncollected for utilization and unoxidized by the top cover of the landfill. The assumptions made for System option 1 are:

- The collection efficiency is 80% during the operational period;
- The oxidation efficiency by top cover is 50% of the methane gas that passes through;
- The volumetric proportion of methane and carbon dioxide at STP is 50%.

In 10 years of operation of a standard LFB and a practical LFG collection efficiency of 80% of the produced LFG the collected and utilizable LFG is 59.3% of the total LFG potential (Figure 7-4). The remaining 40.7% unconverted and uncollected LFG can be regarded as gross loss. However, after the active operation period, all the biodegradable organic matter will be gradually converted into uncollected LFG. Inclusion of an oxidation efficiency by the top cover of 50% yields a net 10.2% (see Figure 7-5) emission of uncollected methane. This net emission of methane is accounted for as contribution to global warming (CH_4-emitted) as shown in Figure 7-5 which is a continuation of Figure 7-4. The CO_2 emissions are the biogenic CO_2 emitted into the atmosphere. This consists of a part of the LFG collected and liberated after utilization of LFG in combined heat and power plants, the uncollected CO_2 and CO_2 stemming from the oxidation of methane in the top cover.

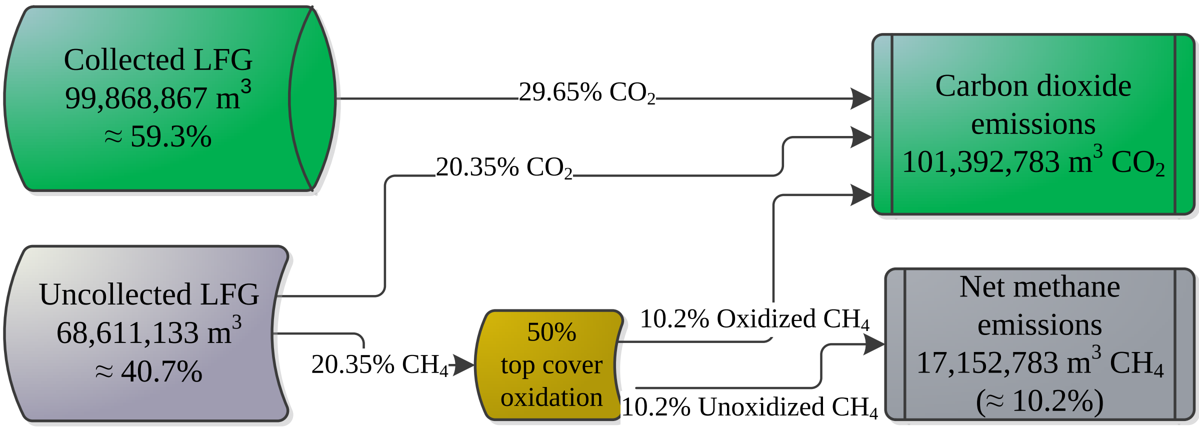


Figure 7-5: Volumes of annual carbon dioxide and net methane emissions for System option 1 for 10 years active operation time (the balance of CO_2 does not include the amount of CO_2 that is produced if the biogas is incinerated)

The biogenic CO_2 emission from the Standard LFB is 101,392,783 m^3 per year as illustrated in Figure 7-5 whilst net methane emission ($CH_{4\text{-emitted}}$) is 17,152,783 m^3 per year (10.2%) being the global warming contribution from this system option. The volume percentages are related to the LFG potential.

CH_4-$_{\text{emitted}}$ = 17,152,783 m^3 CH_4 equivalent to 12,350,004 kg CH_4 (density = 0.72 kg CH_4/m^3)

The global warming factor from emission of LFG (GWF_{LFG}) from the Standard LFB with 10 years of operation is estimated by multiplying the methane emission ($CH_{4\text{-emitted}}$) with a GWP_{CH4} of 21 (IPCC 2001) (i.e.1 kg CH_4 = 21 kg CO_2- eq.).

Accordingly, the global warming factor amounts to:
$GWF_{LFG} = GWP_{CH4} * CH_{4\text{-emitted}}$
= 21* 12,350,004 = 259,350,084 kg CO_2- eq. per year.

This Standard LFB has the potential of contributing GHG to the atmosphere of approximately 260,000 ton CO_2-eq. per 1 million ton (0.26 ton CO_2-eq. per ton) MSW. Upon capture and utilization of LFG carbon credits can be claimed within the Cleaner Development Mechanism (CDM) framework.

Sensitivity Analysis for System option 1

A sensitivity analysis of this system is performed by varying some parameter values and assess the effects on the performance of the system. The same general assumptions made in the previous calculations such as the practical collection efficiency, the CH_4-CO_2 ratio, the oxidation by the top cover, the GWP_{CH4} and leachate recirculation period of 5 years also apply in this sensitivity analysis. The following aspects were studied:

- Effect of the length of active operational time varying between 10 and 200 years and the annual amount of unconverted LFG;
- Effect of the length of operation time varying between 5 and 15 years on the LFG production and collection (80% collection efficiency) and the tonnage of carbon dioxide equivalent that is emitted into the atmosphere;
- Effect of a variation of the collection efficiency in the LFB between 75% and 85% on the amount of LFG collected for a fixed LFG production.

The first parameter discussed is the effect of the operation time of cells being active for a longer period than 10 years and still maintaining the 5 years of recirculation of leachate. We look at the residual potential amount of LFG in the LFB that is still unconverted. For operation times varying from 10 years until 200 years the annual amount and percentage of unconverted LFG potential and amount of fugitive gases produced from 234,000 tons OM are presented in Table 7-4 and Figure 7-6.

Table 7-4: Annual unconverted (not produced) amount of potential LFG and potential CH_4 emission from the unconverted LFG in System option 1 as a function of active period of operation (varying from 10 to 200 years)

Years of active operation	10	15	25	50	100	150	200
Unconverted LFG (x 1,000 m^3)	43,644	33,483	24,806	14,592	5,102	1,612	328
% unconverted of	26%	20%	15%	9%	3%	1%	0.2%

potential LFG							
Potential CH_4 produced (tons)	15,712	12,054	8,930	5,253	1,837	580	118
Potential CH_4 emission (tons)	7,856	6,027	4,465	2,627	918	290	59

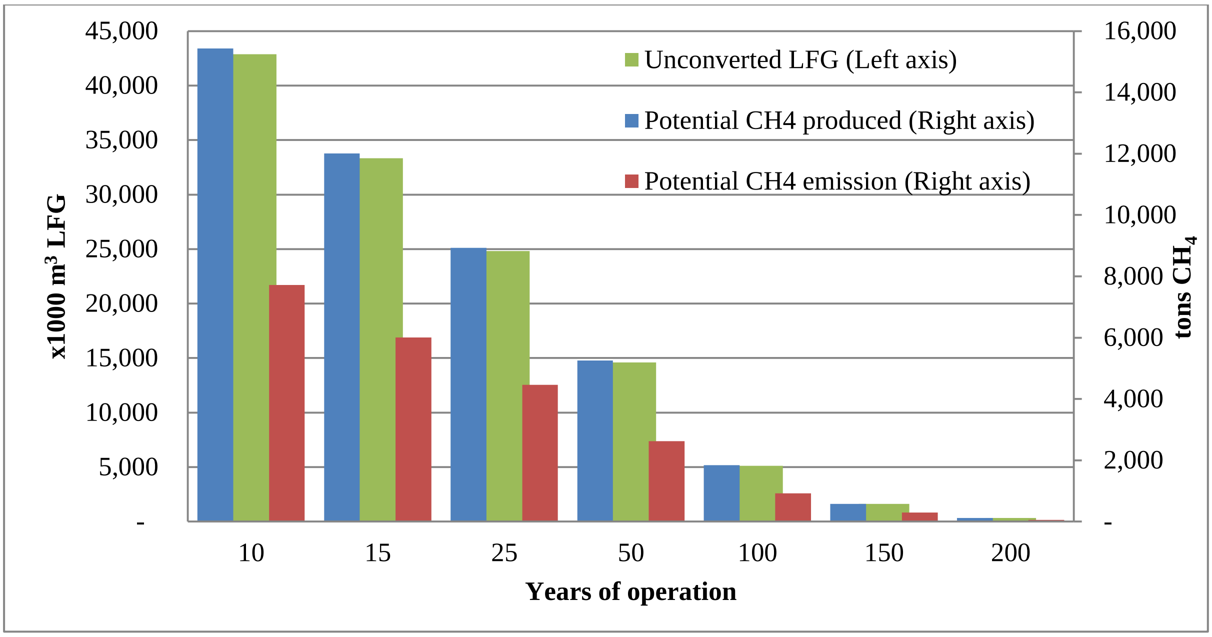


Figure 7-6: Annual unconverted LFG and potential CH_4 emission to the atmosphere by System option 1 as a function of the active period of operation (varying from 10-200 years)

The tonnage and volume of the potential gaseous emissions still left in the landfill site can be deduced from Figure 7-4. After 10 years of operation, 43,644,000 m^3 LFG, which is 26% of the total potential LFG, still remains in the landfill site and there is a gradual decrease in the later years. After 15 years of operation the unconverted LFG is 20% of the initial potential which is 5% less after an additional 5 years of operation. After 25 years of operation, there is still 15% unconverted LFG potential and the methane emission potential has decreased to less than 5,000 tons as the biodegradable organics which produce methane are depleting. After 50 years all the rapidly biodegradable organics will be completely biodegraded. The same is true for most of the moderately biodegradable organics. Some slowly biodegradable organics are still present and the methane potential and consequently the emitted methane is below 3,000 tons which is about 9% of the total LFG potential. After 100 years none of the rapidly and moderately biodegradable organics will be left at the landfill site and the annual potential methane emissions are 3% or less than 60 tons. The time to be selected for operation of the landfill gas recovery is determined by a trade-off between extra operational costs and the benefits from the captured gas. In this thesis a gas recovery period of 10 years is selected.

Table 7-5 summarizes the variation in LFG production and collection (80% collection efficiency), the methane emissions and the corresponding global warming factor as a function of the active operation time (5, 10 and 15 active operational years).

Table 7-5: Annual landfill gas production, methane emission and GWF versus varying active operation times for System option 1 (collection efficiency of LFG: 80%)

Operation time (years)	LFG production (m^3 LFG)	LFG produced (%)	LFG collected (m^3 LFG)	Emitted (m^3 CH_4)	GWF (Ton CO_2-eq.)
5	98,123,052	58.2	78,498,441	22,495,390	340,130
10	124,836,084	74.1	99,868,867	17,152,783	259,350
15	134,997,326	80.1	107,997,860	15,120,535	228,622

A Standard LFB on a 5 years operational time is capable of producing 58.2% of the total LFG potential at a rate of about 98,000,000 m^3 LFG per year and a collectable amount of 78,489,441 m^3. With an additional 5 operational years, in total about 100,000,000 m^3 LFG will be produced corresponding to 74.1% of all OM present in the reactor If the operation time is 15 years, the percentage LFG produced is 6% more than that of 10 operational years with about 7,000,000 m^3 LFG additionally collected.

With respect to the GWF, a reduction of more than 90,000 ton CO_2- eq. can be achieved if the operation period is increased from 5 years to 10 years and a further reduction of 30,000 ton CO_2- eq. after 15 years. It is evident that 10 years is more optimal period of operation than a decade and a half to achieve 80% of the potential LFG production.

Finally, we carried out a sensitivity analysis by varying the collection efficiency of the LFG production of 124,836,084 m^3 over 10 years operational time and assessing the GWF. The results are presented in Table 7-6.

Table 7-6: Effect of change of LFG collection efficiency on annually collected LFG and GWF for System option 1 (operation time: 10 years)

Efficiency (%)	LFG collection (m^3LFG)	GWF (Ton CO_2- eq.)
75	93,627,063	282,944
80	99,868,867	259,350
85	106,110,671	235,756

The influence of changes of the collection efficiency on the collected LFG is relatively small but a larger difference is observed on the GWF. A reduction of 10% collection efficiency from 85% to 75% brings about an extra emission of 50,000 tons CO_2-eq.(about 20% more).

7.2.5 Leachate production and re-circulation

Operation of the standard LFB requires leachate to be collected from the bottom of the reactor and recirculated back into the reactor. For this purpose, leachate should be recirculated using vertical wells. Storage is provided to ensure that peak and off-peak leachate generation events are accommodated. Storage is also important to warrant the supply of leachate for maintenance of moisture content in the LFB at the optimal level of 40 to 50% also referred to as field capacity as discussed in chapter 4.

Leachate production is a function of the amount of water initially present in the waste before landfilling and the amount of water lost from each cell during the active operational period. We assume that after 10 years of operational time the cell is at field capacity. We calculate the amount of leachate that can be produced from the deposited waste at the end of 10 years of active operation on the basis of the initial 234,000 tons dry OM in the deposited 940,000 tons of wet waste. This OM yields 124,836,084 m^3 LFG per year at an average LFG conversion factor of 0.72 m^3 LFG/ kg OM (section 7.2.4). The amount of organics converted to produce the 124,836,084 m^3 LFG is $\frac{124{,}836{,}084}{0.72}$ = 173,384 tons. So, after 10 years of active operation 234,000 - 173,384 = 60,616 tons OM is still left in the landfill site. The remaining total solids which is the sum of the biodegradable organics still present (60,616 tons) and 66,000 tons inert organics and inorganics (Table 7-1) which amounts to 126,616 tons of dry solids. At field capacity the water content is 0.4 times the amount of solids still present which means that the amount of water still in the waste is 0.4 * 126,616 = 50646 tons water. As the initial water content in the 940,000 tons of wet waste was 640,000 tons, the amount of water released from the waste is 640,000 – 50646 = 589,354 tons of water per year. Therefore, the amount of leachate is approximately 589,354 tons/year, if other possible additions or abstractions of water such as rainfall and evaporation are neglected. This leachate is used for recirculation and needs a final treatment at the landfill site.

Recirculation

Leachate recirculation schedules are set in response to leachate and gas monitoring results so that leachate recirculation is in harmony with the progress and intensity of stabilization. Leachate from older cells mixed with fresh leachate is used to provide start-up seeding for newer lifts.

The leachate recirculation rate will be such that the leachate is recirculated into the waste mass will reach the bottom within at least one year. Given the cell height of 10 m and an assumed waste porosity of 40%, the liquid surface loading rate would be about 10 mm/day (10 l/m^2.day or 100 m^3/ha.day).

The leachate not required for recirculation requires post-treatment and disposal. Figure 7-7 is an illustration of the water balance of the LFB, the leachate storage and recirculation, and the leachate treatment. The quantity of leachate that needs post-treatment (Q_j) is the excess leachate that remains after fulfilling the recirculation requirement (Q_g).

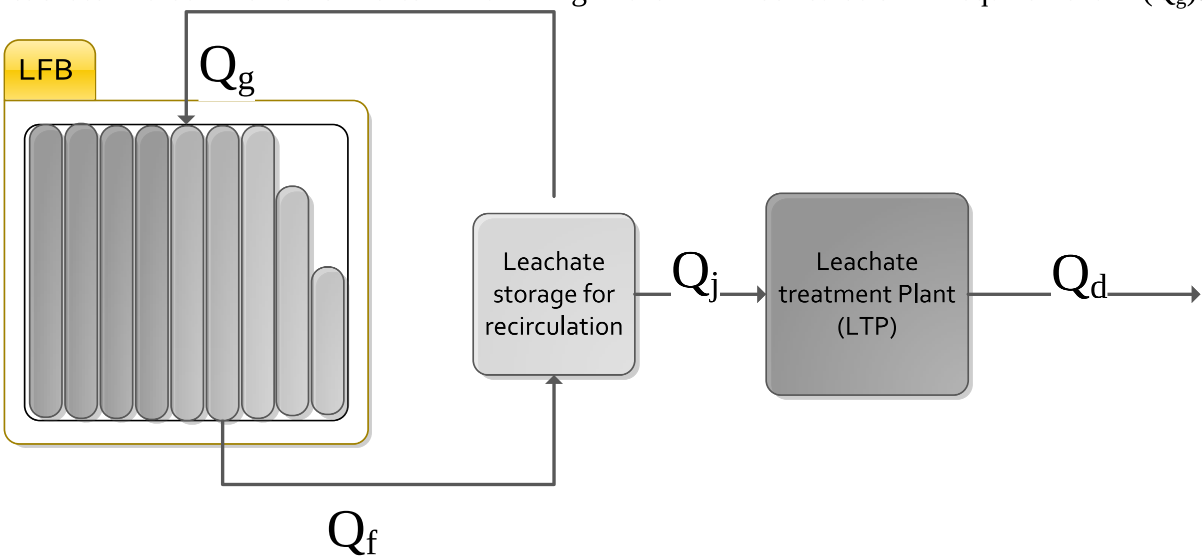


Figure 7-7: Water balance for System option 1- Standard LFB

From Figure 7-7:
Q_f Collected leachate;
Q_g Recirculated leachate;
Q_d Effluent from LTP;
Q_j Discharge of excess leachate to LTP.

The leachate treatment plant removes mainly COD and nutrients from the excess leachate before the effluent (Q_d) is discharged into receiving water bodies or municipal sewers. When necessary, pH adjustment is done in the leachate storage tank to avoid extended periods of acidification in the LFB as was observed during the pilot-scale experiment presented in chapter 6. If $Q_f < Q_g$, the quantity of leachate for recirculation is too low and additional water is required to maintain the recirculation rate. Treated leachate from the LTP can be used as additional recirculation water.

The recirculation regime follows the pattern of the opening and closing of cells described in section 7.2.3 and Figure 7.3. Recirculation is restricted to the fully active cells only during the first 5 years of operation. Thereafter, no recirculation of leachate is carried out. After 5 years the recirculation system is uncoupled but the LFG and leachate collection continues for another 5 years as shown in Figure 7-8.

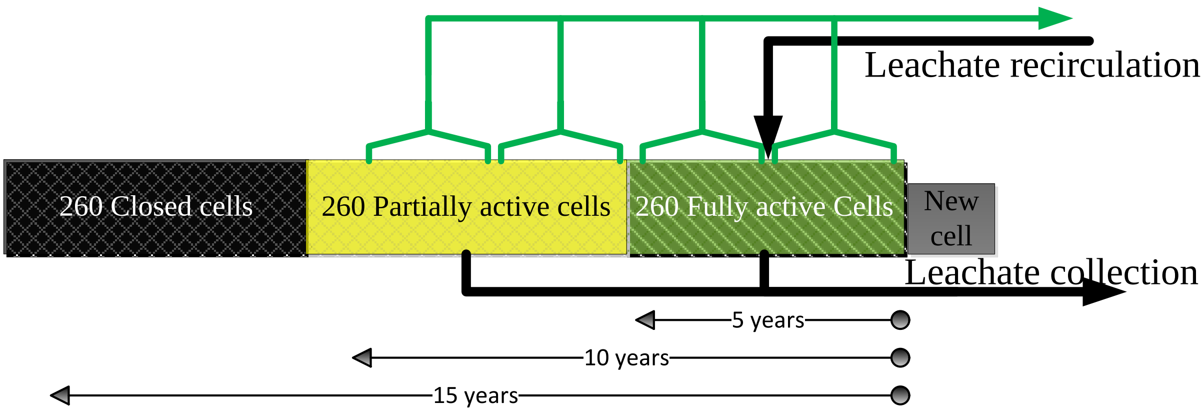


Figure 7-8: Leachate recirculation regime

7.2.6 Leachate treatment

Treatment of leachate is a dual process that takes place within the bioreactor during the recirculation and during post-treatment in a LTP to ensure the leachate is safe for disposal. Here, under the assumptions of table 7-1, the amount of the leachate to be treated is 589,354 tons/year (589,354 m^3/year). The composition of the recirculated leachate that requires post treatment has been discussed in chapter 4. Post-treatment of excess leachate is imperative because pollutants such as non–biodegradable COD, ammonia-nitrogen and other pollutants are insufficiently removed during recirculation. The proposed leachate treatment options are presented in chapter 4 of this thesis.

7.3 System option 2: STANDARD LFB WITH ACIDIFICATION OF LEACHATE AND LFG PRODUCTION IN A SEPARATE REACTOR

7.3.1 General layout of option 2

System Option 2 comprises of the Standard LFB (discussed in section 7.2.3) coupled with acidified leachate treatment and LFG production and collection in a separate UASB reactor as illustrated schematically in Figure 7-9. It is assumed that the biogas from the UASB reactor and the LFG generated in the landfill both have a composition of 50% CH_4 and 50% CO_2.The system option has two recirculation circuits which shall be discussed.

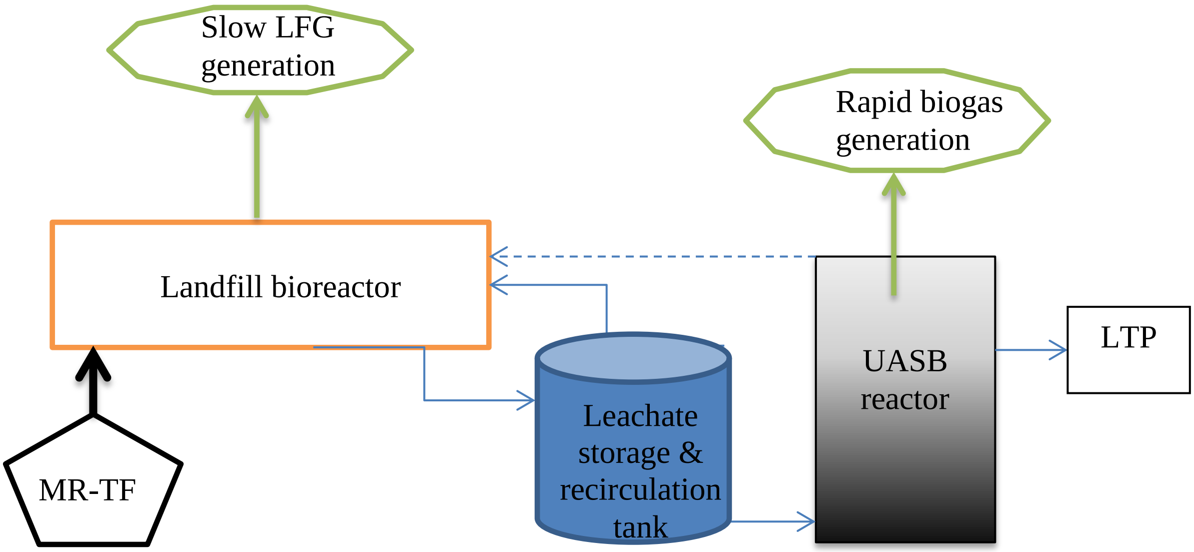


Figure 7-9: System option 2 - Standard LFB with acidification of leachate and LFG production in a separate UASB reactor

7.3.2 Materials Recovery and Transfer station Facility

MSW collected from waste producers is transported to transfer stations. The MR-TF for this option is similar to the one described for Standard LFB - option 1 with a flow diagram as already discussed in section 7.2.2 and Figure 7-2. From the transfer stations, the sorted MSW is transported to the disposal site and filled in the LFB. The amount and characteristics of waste received at the station and the composition of recyclables and reusables removed from the waste stream are similar to that of System option 1.

7.3.3 Landfill Bioreactor

The bioreactor is a Standard LFB that receives sorted waste from MR-TF. Filling of the cells of the bioreactor follows the same protocol as described in Figure 7-3. The bioreactor in this option is partitioned in terms of the age of cells. The partition of old and new cells is in accordance with the recirculation of leachate to intensify an acidification phase in the new cells. The acidification regime proposed lasts one year during which only recirculation and collection of leachate takes place. In that one year all the water initially present in the waste is replaced at least one time by acidified water. This means all leachate in the waste becomes acidified. After the first year each week a new cell is connected to the storage and recirculation of the acidified leachate. Connecting a new cell each week also means that each week a cell with an age of 1 year is disconnected from the acidified leachate recirculation system. During the acidification regime there is hardly any LFG generation in the landfill. The acidified leachate is passed from the LFB to a methanogenic UASB reactor in order to rapidly generate biogas and the effluent of this reactor is recirculated over the non-acidified cells. After 1 year the new cells are categorized as old cells and normal recirculation of leachate and gas collection for 4 years is carried out on these cells and then during an additional 5 years the LFB is operated without leachate recirculation but with collection of gas following the pattern described in Figure 7-8. In the meantime newer cells continue to be recirculated with acidified leachate. The closure and opening of cells follows the same pattern as for System option 1.

This system option receives 940,000 tons/year of waste received from the MR-TF and the time taken to fill each cell is one week just like for System option 1. About 18,000 tons of waste per week shall be deposited at the landfill site. The surface area required to fill the 18,000 tons of waste with density of approximately 0.5 ton/m^3 and 10 m high is 18.8 ha/year similar to the land requirement for System option 1.

7.3.4 LFG generation in System option 2

LFG potential

In this paragraph we discuss the production of biogas and the greenhouse gas emissions independently from the leachate collection, recirculation and final treatment system. We assume that an appropriate collection, recirculation and treatment system is sufficiently robust and it has no influence on biogas production as it was the case for System option 1.

Landfill gas generation in this option occurs in two separate reactors. A part of LFG is envisaged to be generated in the UASB reactor as biogas and another part in the standard LFB as LFG. As in System option 1, LFG is generated from biodegradation of the 234,000 tons per year dry OM in the wet waste. It is envisaged that the composition of the gas from both reactors will be 50% CH_4 and 50% CO_2 v/v at STP. The LFG potential is the same as that of System option 1 namely 168,480,000 m^3 LFG. The intention here is that leachate is recirculated for one year under an acidification regime whereby the production rate of biogas in the UASB is comparable with the production from the LFB as used in System option 1. The difference is the UASB system exhibits no loss of biogas.

LFG production

The amount of biodegradable organic waste reaching the disposal site in this system option is 234,000 tons/year (Table 7-1). It is assumed that the rate of acidification in this System option is equal to the rate of biodegradation in a Standard LFB. The acidification step including hydrolysis is the rate determining step in the production process of biogas. The rate of conversion of VFA present in the acidified leachate into biogas in the UASB is fast but the biogas production during and after the acidification regime is the same as System option 1 which amounts to 124,836,084 m^3 LFG/year for 10 years of active operation. Using Equation 5-12 derived in chapter 5, the biogas production from the UASB during the 1 year of acidification of the leachate is 32,494,214 m^3 biogas equivalent to 19.3% of the potential LFG production. Thereafter LFG is continued to be produced from the older cells of the LFB. So for the remaining 9 years of the LFB's 10 years of active operation and LFG collection operation the LFG produced is 124,836,084 – 32,494,214 = 92,341,870 m^3 LFG/year equivalent to 58.4% of the potential gas.

The gas production from this system is therefore the sum of the biogas generated in the UASB (32,494,214 m^3 biogas) and in the LFB (92,341,870 m^3 LFG) amounting to 124,836,084 m^3 LFG or 74.1% of the total potential. That leaves after 10 years 168,480,000 – 124,892,281 = 43,643,916 m^3/year potential LFG unconverted which is exactly the same as for System option 1.

Practically, the LFG collection efficiency is 80% at the LFB and 100% at the UASB reactor. So from the LFB in combination with the UASB reactor the amount that can be collected within the 10 years of operation is 106,367,710 m^3 LFG/year (≈63.1% of the LFG potential). 62,112,290 m^3 LFG/year remains uncollected but a portion (50%) of the methane gas in it is subject to oxidation by the top cover and can be accounted for as avoided emissions (Fig 7-10).

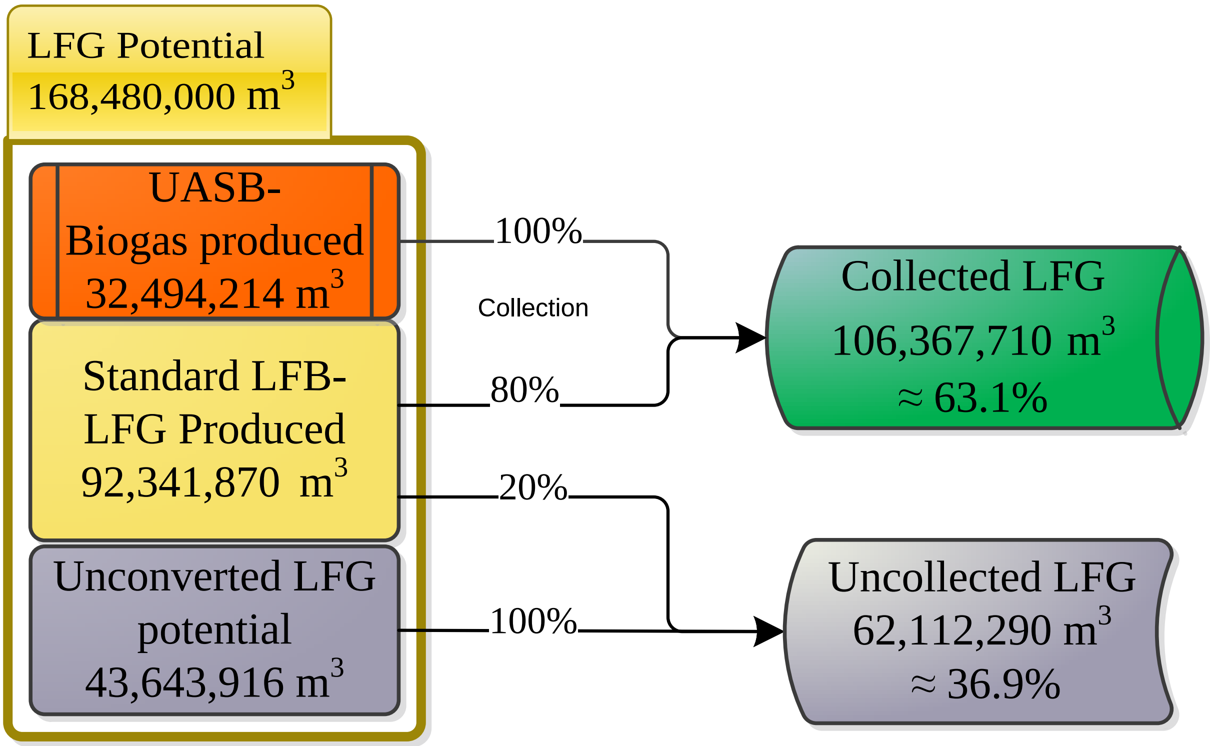


Figure 7-10: System option 2 – Volume of collected and uncollected LFG during 10 years operational time

Estimation of greenhouse gas emission

Methane from the UASB reactor and the LFB is considered to be a GHG, but carbon dioxide is not as it is in System option 1. Carbon dioxide emission from the waste mass is considered biogenic and as such does not contribute to the GWF. Estimation of greenhouse gas emission associated with Standard LFB coupled with LFG production in a separate UASB reactor is based on the following assumptions:

- No greenhouse gas emission from the UASB because collection is 100%;
- Very small amount of GHG emission during 9 years of operation of the LFB due to uncollected LFG. The practical LFG collection efficiency at the LFB is 80%;
- During and after 10 years of operation the typical 50% oxidation efficiency of the methane gas that passes the top cover.

In a span of 10 years of operation the Standard LFB coupled with a UASB reactor is capable of converting about 74.1% of the available biodegradable organic matter into LFG. With 100% collection efficiency in the UASB reactor and a practical LFG collection efficiency of 80% at the LFB the collected LFG will amount to 63.1% of the LFG potential. The remaining 36.9% LFG which is unconverted and uncollected within the 10 years of operation time can be regarded as gross loss. However after the active operation period, all the remaining biodegradable organic matter will gradually be converted to uncollected LFG. Taking into account a typical oxidation efficiency by the top cover of 50% the net emission of uncollected methane, unoxidized by the top cover is 15,516,833 m^3 CH_4 (9.2% of the LFG potential). This counts as the contribution to global warming (CH_4-emitted) revealed in Figure 7-11. Carbon dioxide emissions are 99,756,833 m^3 CO_2 which is partly the collected and uncollected gas during the 10 years of operation as well as the gas emitted after closure of the LFB. Since this gas is biogenic it is not considered in the GHG accounting.

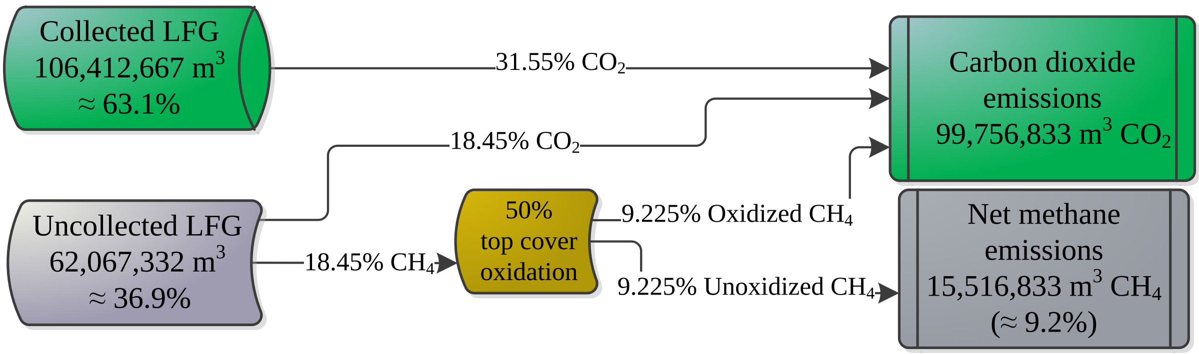


Figure 7-11: Volumes of carbon dioxide and net methane emissions for System option 2 for 10 years active operation time

The greenhouse gas of concern is the methane that can be emitted ($CH_{4\text{-emitted}}$) into the atmosphere which amounts to 9.2% of the LFG production (Figure 7-11).

The $CH_{4\text{-emitted}}$ = 15,516,833 m^3 CH_4 equivalent to 11,172,120 kg CH_4 (density of methane = 0.72 kg/m^3).

The global warming factor based on the emission of LFG (GWF_{LFG}) from this system option is estimated by multiplying the methane emission ($CH_{4\text{-emitted}}$) with a GWP_{CH4} of 21 (i.e.1 kg CH_4 = 21 kg CO_2- eq.).

Accordingly, the global warming factor is calculated as follows:

$$GWF_{LFG} = GWP_{CH4} * CH_{4\text{-emitted}}$$
$$= 21 * 11{,}172{,}120 = 234{,}614{,}520 \text{ kg } CO_2\text{- eq. per year}$$

This Standard LFB coupled with a UASB reactor has an annual contribution of methane emission into the atmosphere of approximately 234,615 ton CO_2-eq per 1 million tons MSW (0.23 ton CO_2-eq per ton). Upon capture and utilization of LFG for electricity generation and within the Cleaner Development Mechanism (CDM) framework carbon credits can be claimed for the avoided methane emissions.

Sensitivity Analysis for System option 2

A sensitivity analysis of this system is performed by varying some parameter values and assess the effects on the performance of the system. The same general assumptions made in the previous calculations such as the practical collection efficiency, CH_4-CO_2 ratio, the oxidation by top cover, the GWP_{CH4} also apply in this sensitivity analysis. The following aspects were studied :

- Effect of the duration of the period of active operation on the annual amount of unconverted LFG. This duration was varied from 10 to 200 years;
- Effect of the duration of the period of active operation on the annual landfill gas production, methane emission and GWF. This duration was varied between 5 and 15 years;
- Effect of the LFB collection efficiency varied between 75% and 85%;
- Effect of the duration of the acidification regime from 0 years to 2 years at a 10 years active operation time and 80% collection efficiency.

For landfill operation periods varying from 10 years until 200 years the amount and percentage of unconverted LFG potential and amount of fugitive gases are the same as found for System option 1, as could be expected on the basis of the assumptions made.

The effect of the duration of the period of operation (for a acidification period of 1 year) on the LFG production and collection (80% collection efficiency), and the tonnage of carbon dioxide equivalent that is emitted into the atmosphere, is given in Table 7-7.

Table 7-7: Annual landfill gas production, methane emissions and GWF for System option 2 as a function of active period of operation (Collection efficiency of LFG: 80%)

Operation time (years)	**LFG production** (m^3LFG)	**LFG produced** (%)	**LFG collected** (m^3LFG)	**Emitted** (m^3 CH_4)	**GWF** (Ton CO_2- eq.)
5	98,123,052	58.2%	84,997,284	20,875,608	315,639
10	124,836,084	74.1%	106,367,710	15,528,073	234,784
15	134,997,326	80.1%	114,496,703	13,492,234	204,003

A Standard LFB coupled with a UASB reactor at a 5 years operational time is capable of producing 58.2% of the total LFG potential at a rate of about 98 million m^3 LFG per year. After an additional 5 years, almost 74.1% of all OM present in the reactor is converted to LFG. If the operation time is extended to 15 years, an additional 6%.of LFG is produced. As for the GWF, a reduction of 80,000 respectively 30,000 ton CO_2- eq. can be achieved if the operation period in prolonged from 5 to respectively 10 and 15 years. It is evident that an operation time of 10 years would be a more optimal time than a short 5 years or a decade and a half to achieve 80% LFG production.

The effect of the variation in the efficiency of the collection of the 124,836,084 m^3 LFG s produced in the 10 years operational time on the collected amount and on the GWF is presented in Table 7-8.

Table 7-8: Effect of variation of LFG collection efficiency on collected LFG and GWF for System option 2 (operation time 10 years, acidification period 1 year)

Efficiency (%)	**LFG collected** (m^3LFG)	**GWF** (Ton CO_2- eq.)
75	101,750,616	252,237
80	106,367,710	234,784
85	110,984,803	217,332

A 5% increase of the efficiency of collection results in about 5 million m^3 LFG additionally collected and a significant reduction of the GWF is observed. A decrease of the collection efficiency by 10% (from 85% to 75%) brings about an extra methane emission of 35,000 tons CO_2-eq. This is 16% of the GWF at 85% collection efficiency.

The fourth factor that has been studied concerns the acidification period. This period was varied at four levels: 0, 0.5, 1 and 2 years. The effect of the length of the acidification period on the LFG produced and collected, the potential methane emissions and the GWF is summarized in Table 7-9.

Table 7-9: Effect of change of the duration of the acidification regime on LFG production and collection, potential methane emission and GWF for System option 2 (80% collection efficiency, operation time 10 years)

Acidification regime (years)	**LFG production** (m^3 LFG)	**LFG collected** (m^3 LFG)	**Emitted** (m^3 CH_4)	**GWF** (Ton CO_2- eq.)
0	124,836,084	99,868,867	17,152,783	259,350
0.5	124,836,084	103,370,470	16,277,383	246,114
1	124,836,084	106,367,710	15,516,833	234,615
2	124,836,084	111,156,436	14,330,891	216,683

It is evident from Table 7-9 that the duration of the acidification regime has no impact on annual LFG production but it has an impact on the amount of LFG that can be collected by System option 2. A prolongation of the acidification regime from 1 to 2 years brings about 5 million m^3 per year more collected LFG and consequently 18,000 tons CO_2- eq. per year reduction of potential methane emission to the atmosphere. A change of the acidification regime from 0 to 0.5 years and from 0.5 to 1 year yields an additional amount of LFG collected of 4 million m^3 and 3 million m^3 LFG respectively. However, it should be noted that to maintain acidification in the LFB for 2 years may not be very realistic.

7.3.5 Leachate production and recirculation

Leachate production for this option is similar to that of System option 1 as discussed in section 7.2.5. The amount of leachate that can be produced from the deposited waste is 589,354 tons water/year (589,354 m^3 water /year).

Recirculation

In this system option, acidification of the leachate is crucial. Therefore leachate from older cells is not mixed with fresh (young) leachate during recirculation. Instead the fresh leachate is recirculated for one year of operation in the new cells only. The leachate recirculation rate for both the old and new cells is the same and will be such that the leachate recirculated back to the waste mass will reach the bottom within at least one year. Given a cell height of 10 m similar to the cells of the Standard LFB (system option 1) and an assumed porosity of 40%, the liquid surface loading rate would be at least 10 mm/day (10 l/m^2.day or 100 m^3/ha.day) similar to that of System option 1.

Standard LFB operation requires leachate to be collected from the bottom of the reactor and recirculated back into the reactor as described for System option 1. Young and old leachates are collected and stored in separate tanks as depicted in Figure 7-12. Leachate from older cells (Q_f) is recirculated to old cells only (Q_g) and is strictly prohibited for newer cells. The young leachate (Q_c) from the newer cells is recirculated within the newer cells only (Q_a) so as to have an extended period of 1 year of acidification of the leachate. Taking new cells in operation means disconnecting cells from the acidified recirculation and coupling them to the non–acidified recirculation. Given the landfill surface area containing the fresh waste deposited over one year of 18.8 ha, the flow to be recirculated is 18.8 * 100 = 1880 m^3/d or about 686,200 m^3/yr. Hydrolysis and acidification are occurring in the LFB simultaneously. The initial deposition of fresh waste and a high recirculation rate of leachate causes a drop of pH to nearly 5 as was observed during the pilot scale LFB study (chapter 6). The leachate can be kept acidic as long as fresh waste is continuously filled in new cells and this leachate is collected and recirculated for 1 year. The excess acidified leachate (Q_b) is fed into the methanogenic UASB reactor for rapid generation of biogas and pre-treatment. Leachate fed into the UASB for biogas production (Q_b) = collected leachate from new cells (Q_c) minus the amount of acidified leachate recirculated to new cells (Q_a). Excess leachate from the older cells, (Q_j), and from the UASB reactor (Q_e) is diverted to a leachate treatment plant (LTP) for post-treatment. If the old cells do not generate enough leachate, additional water is needed.
This additional water could be treated leachate from the UASB reactor and/or from the LTP. But if additional water is needed for new cells, it can only be drawn from the LTP and not from the UASB reactor. Leachate for final treatment is the sum of excess leachate (Q_j) and UASB effluent (Q_e). Treated leachate (Q_d) discharged from the LTP shall be released into receiving water bodies or municipal sewers.

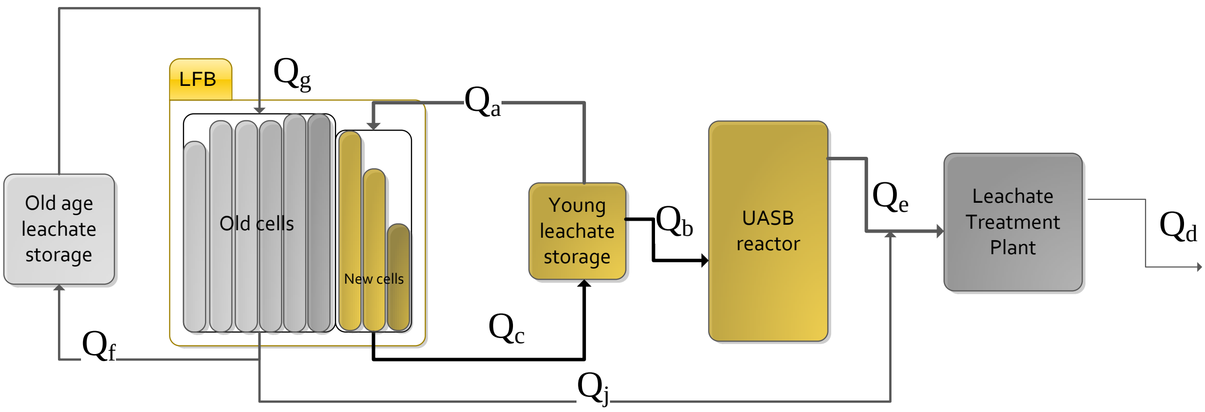


Figure 7-12: Water balance for System Option 2

From Figure 7-12;

Q_a Recirculation of acidified leachate to new cells;
Q_b Acidified leachate for UASB treatment and biogas production;
Q_c Collected leachate from new cells;
Q_d Effluent from leachate treatment plant;
Q_e Effluent of UASB reactor;

Q_f Recirculated leachate from older cells;
Q_j Excess leachate.

7.3.6 Leachate treatment

Leachate in this option undergoes treatment in the following sequence. First young leachate is recirculated back into the waste mass of the young LFB cells. Surplus leachate is treated in the UASB reactor for one year during the acidification regime. Secondly, after one year the young leachate is recirculated to the LFB's older cells and any excess leachate is then diverted to a LTP for post-treatment before discharge into receiving water. The leachate treatment options are discussed in chapter 4 and are mainly geared to remove the remaining organic matter and nutrients.

7.4 System option 3: TWO STAGE TREATMENT – CENTRALIZED BIOCEL AND LFB

7.4.1 General layout of option 3

This system option is a two stage treatment process whereby waste is pre-treated biologically by means of anaerobic digestion in a centralized BIOCEL-reactor located close to the LFB. The pre-treated waste is then moved into the LFB as illustrated in Figure 7-13 and conventional Standard LFB processes are applied. It is assumed that the rapid biogas and slow LFG generated have the same composition of 50% CH_4 and 50% CO_2.

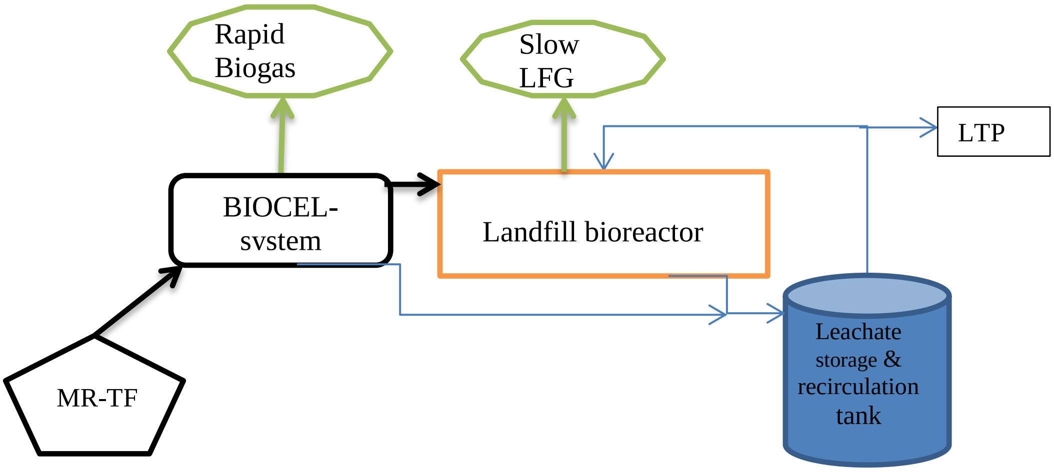


Figure 7-13: System option 3 - Two stage treatment – Centralized BIOCEL-system and LFB

7.4.2 Materials Recovery and Transfer station Facility

The MR-TF proposed for this option is similar to the ones described in the preceding sections whereby packed and unpacked commingled and source separated wastes (present and future) are discharged in a receiving area. At the receiving area sorted packages known to contain biodegradable organics are directly transported to the BIOCEL reactors while reusable and recyclables are taken to recycling streams. The same happens for unpacked commingled waste contained in bags and sacks which are then opened up, subjected to separation of recyclables after which biodegradables are transported to the landfill site. The characteristics and composition of waste reaching the MR-TF is similar to that of the previous options and shown in Table 7-1.

7.4.3 BIOCEL-system

The BIOCEL-system at the landfill site is the first to receive sorted waste from the MR-TF for anaerobic digestion of the waste under controlled conditions. The BIOCEL process as described in chapter 3, is an anaerobic digestion technology for MSW organic fraction based on a batch-wise digestion at high solid concentrations at mesophilic temperature. The anaerobic digestion is carried out in concrete digesters where waste is kept approximately 20 days and we assume that in these 20 days only the rapidly biodegradable organics are converted into biogas. During the anaerobic process leachate is recirculated in the BIOCEL reactors. The floors of the digesters are perforated at the bottom for leachate collection. The conversion of rapidly biodegradable OM into biogas results in reduction of the amount of organics originally present in the waste which has an impact on the overall amount of waste to be landfilled. We assume that field capacity cannot be achieved at the MR-TF stations, or during operation of the BIOCEL reactor. The assumption is that, the water fraction of the waste that enters the BIOCEL-system is equal to the water fraction of the waste that leaves the BIOCEL-system. As the amount of organic matter decreases, a small amount of surplus leachate is produced in the BIOCEL operation and is included in the recirculation system of leachate. Therefore we can calculate the amount of surplus leachate that is released during BIOCEL operation and the volume/weight that has to be disposed in the LFB.

The capacity of the reactors is aimed to be able to accommodate at least 2500 tons/day (940,000 tons/year) i.e. the daily amount of waste collected and sorted at the MR-TF and kept for at least 20 days in the BIOCEL. Seeding of fresh waste with treated waste is necessary. This means that the amount of waste in the BIOCEL reactors is more than the 2500 tons/day. We assume that the seed material on a weight basis is one third of the fresh waste, this is 0.33*2,500 =825 ton/day. This means that the total amount of material fed into the BIOCEL reactors = 2,500 + 825 = 3,325 tons/day. It is assumed that no additional amount of biogas is produced from the seed material . After 20 days the BIOCEL reactors are opened and one third of the material is reused as seed material and the remaining material goes to the landfill bioreactor cells. The density of the mixture of seed material and fresh waste that goes to the BIOCEL reactors is 0.5 ton/m^3. For a 4 metre waste working height and 29 by 29 sq. metres reactor then at least 40 such reactors should be able to pre-treat waste in the BIOCEL-system. The land requirement for the reactors alone is 841 m^2 * 40 reactors = 33,640 m^2 or 3.364 ha.

As already mentioned one of the advantages of the BIOCEL-system is the ability to collect 100% of the biogas produced and a higher conversion rate due to improved process control. According to a mass balance performed by ten Brummeler (1993), in a BIOCEL-system 1 kg VS (corresponding with an amount of biodegradable OM of about 1 kg) yields 0.25 kg CH_4 and the BIOCEL-system is capable of producing 70 kg biogas per ton biowaste. On the basis of 50% CH_4 and 50% CO_2 in biogas 0.25 kg methane corresponds with 44/16 * 0.25 = 0.69 kg carbon dioxide. Thus 1 kg of OM converted produces 0.25 kg methane + 0.69 kg carbon dioxide = 0.94 kg biogas. We therefore adopt that 1 kg biogas corresponds to 1 kg rapidly biodegradable OM converted. However, ten Brummeler did not indicate the characteristics of the biowaste used to establish the 70 kg biogas production per ton biowaste. We assume that this biowaste contained partly water, inorganics and 300 kg OM / ton wet waste.

The amount of waste to be transported and disposed at the landfill site from the BIOCEL-system is proportional to the amount of (dry) organics which is reduced by a factor to 70/300 = 0.233 (i.e. 70 kg biogas is produced from 300 kg biodegradable OM/) and 0.767 is the fraction of the remaining organics. After 20 days in the BIOCEL-system from the initial 234,000 tons/year of dry biodegradable OM the remaining biodegradable OM is 0.767 * 234,000 ≈ 180,000 tons OM per year. The sum of inert organics and inorganics (66,000 tons/year) which remains unchanged plus the remaining OM is the total amount of dry solids remaining = 180,000 + 66,000 = 246,000 tons dry solids per year. Here the assumption is that, the waste leaving the BIOCEL-system is not at field capacity but has the same water content as the waste entering the BIOCEL reactors. The initial amount of water in the waste was 640,00 tons/year and 300,000 tons of waste. With 246,000 tons solids the amount of water still in the waste at the same percentage is 524,000 tons/year. Therefore the amount of waste to be transported to the LFB is 246,000 ton solids + 524,000 tons water = 770,000 tons wet waste/year.

7.4.4 Landfill Bioreactor

The LFB in this option receives pre-treated waste from the BIOCEL reactors with most of the rapidly biodegradable organic matter already converted into biogas in the BIOCEL-system. The amount of waste to be landfilled is 770,000 tons/year which makes the land requirement of the LFB for this option to be lower than that of System option 1. The filling protocol of the cells of the bioreactor landfill is similar to that described in Figure 7-3 and follows the same pattern of closure and opening of cells as System option 1. The difference is that it will take longer to fill one cell of the same size as of System option 1 and finally less cells are needed or the cells will be much smaller than the cells in System option 1. For a cell filled with 10 m high waste with a density of 0.5 ton/m^3 the total surface area required is 154,000 m^2/year, equivalent to 15.4 ha per year.

7.4.5 LFG generation in System option 3

LFG potential

As for System options 1 and 2, we discuss in this paragraph the production of biogas and the greenhouse gas emissions independently from the leachate collection, recirculation and treatment system. We assume that the collection, recirculation and treatment system is sufficiently robust and it has no influence on biogas production.

Landfill gas generation in this option occurs in two separate reactors. The reactors for this option are the BIOCEL-system and the Standard LFB. Rapid biogas generation is envisaged first in the BIOCEL reactor during about 20 days and then from the standard LFB at a relatively slow rate as LFG over the 10 years expected life span. Similarly, this System option has exactly the same gas production potential as System option 1 which is 168,480,000 m^3 LFG and the gas composition from both the reactors is the same at 50% CH_4 and 50% CO_2 v/v at STP. The envisaged collection efficiency of the produced LFG is 100% in the BIOCEL-system and 80% in the LFB.

LFG production

Taking the biogas production at 70 kg per 300 kg dry OM then from the 234,000 tons biodegradable OM/year that reaches the BIOCEL will produce LFG equal to $\frac{234,000}{0.3} * 70 =$ 54,600,000 kg biogas/year or 54,600 tons biogas equivalent to 40,444,444 m^3 biogas (density of biogas = 1.35 kg/m^3 at STP). This is equal to 24% of the biogas potential of the waste that is disposed per year. We assume that in the BIOCEL system only rapidly biodegradable organic matter is converted to biogas. In addition to waste reduction, the BIOCEL-system is also characterized by a 100% biogas collection efficiency and significant biogas production in only 20 days.

After the treatment in the BIOCEL, the partially biodegraded waste is then filled into the LFB. LFG production from 10 years of operation from the remaining amount of biodegradable OM is calculated using Equation 5-12 from chapter 5 but with a correction of the OM content to account for the for portion already biodegraded by the BIOCEL-system. The initial amount of waste M_o was 940,000 tons/year. From the equation X_o, k and f values for the organic fractions slowly, moderately and rapidly biodegradable are obtained from Table 7-2. To take care of the reduction of remaining OM due to the BIOCEL activity, equation 5-12 is modified as shown in equation (7-1). With this equation the annual amount of biogas (Q) produced in the LFB is given by:

$$Q = \left[M_o * X_{o,s} * f_s * (1 - e^{-k_s * t})\right] + \left[M_o * X_{o,m} * f_m * (1 - e^{-k_m * t})\right] + \left[(M_o * X_{o,r} - M_{o,b} * X_{o,b}) * f_r * (1 - e^{-k_r * t})\right] \ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots . \ldots\ldots .. (7-1)$$

Here, the amount of rapidly biodegradable OM converted into biogas in the BIOCEL-system is given by: $M_{o,b} * X_{o,b}$.

Therefore the amount of rapidly biodegradable OM still present for landfilling is given by the difference between the initial rapidly biodegradable OM and the rapidly biodegradable OM converted in the BIOCEL-system which is given by:
$M_o * X_{o,r} - M_{o,b} * X_{o,b}$.

The assumption has been made that 1 kg biogas is produced from 1 kg rapidly biodegradable OM converted. Consequently, the 54,600 ton biogas generated in the BIOCEL-system corresponds to 54,600 tons of rapidly biodegradable OM converted.

Using the adapted equation (7-1) the amount of LFG that is generated per year by the LFB during 10 years operational time of the waste in the LFG becomes 85,172,109 m^3 LFG (i.e. 51% of the potential generation of LFG).

From this calculation it can be concluded that the total LFG production per year of this option from the BIOCEL-system (40,444,444 m^3) and for the 10 years operational time of the LFB (85,172,109 m^3) is 125,616,554 m^3 which is ≈75% of the LFG potential. This leaves 168,480,000 – 125,616,554 = 42,863,446 m^3 potential LFG still unconverted.

Practically, it is assumed that the generated LFG collectable from the BIOCEL-system is 100% and 80% from the LFB. From the LFB in combination with the BIOCEL-system the annual amount of LFG that can be collected based on a 10 years operational time as presented in Figure 7-14 is 64.4% (108,582,132 m^3) of the LFG potential. The remaining 35.6% (59,897,868 m^3) is deemed uncollected LFG but a portion of it is subject to oxidation by top cover so can later be accounted for as gaseous emissions avoided.

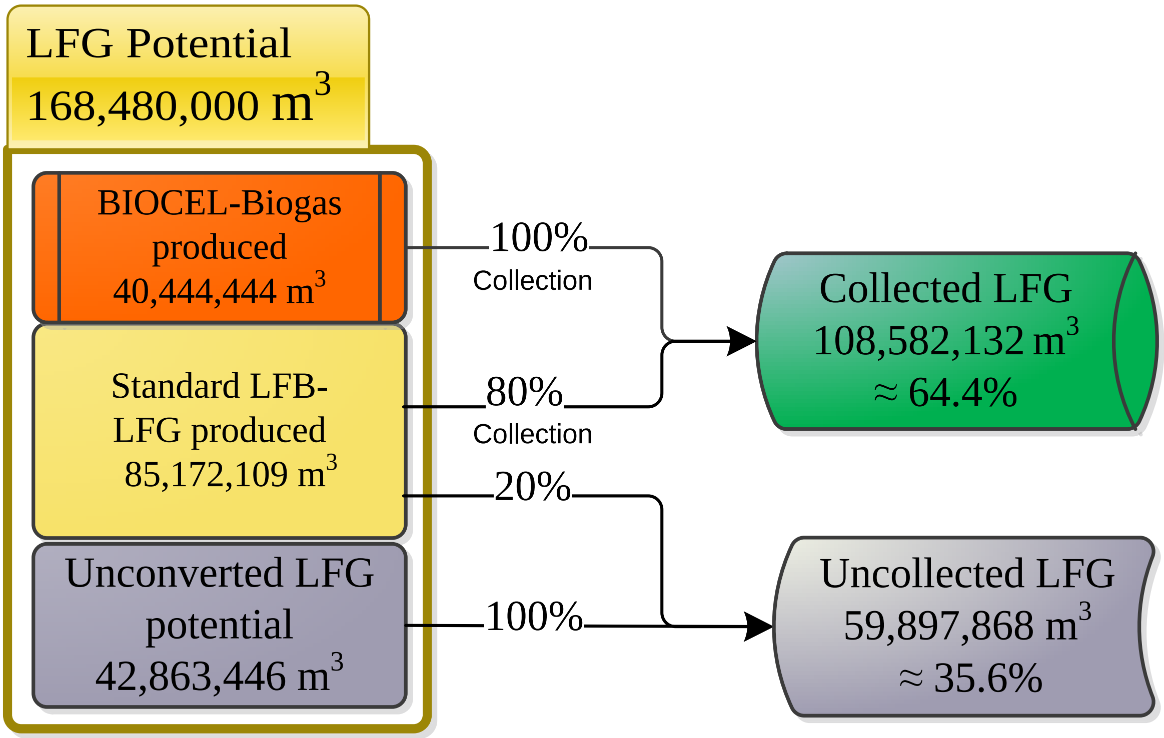


Figure 7-14: System option 3 – Volumes of collected and total uncollected LFG for a 10 years operational period of the LFB

Estimation of greenhouse gas emission

Methane emission from landfills is one of the major contributors to the Greenhouse effect in the world. As already mentioned in section 7.2.4, methane directly from the LFB is considered to be a GHG, but carbon dioxide is not. C

Estimation of greenhouse gas emission associated with BIOCEL-system and Standard LFB is based on the following:

- No greenhouse gas emissions from BIOCEL-system because collection is 100%;
- Small amount of greenhouse gas emission from the LFB during 10 years of operation due to uncollected LFG (Practical LFG collection efficiency at the LFB is 80%);
- During and after 10 years of operation the typical oxidation efficiency of the top cover is 50% of the methane gas that passes through.

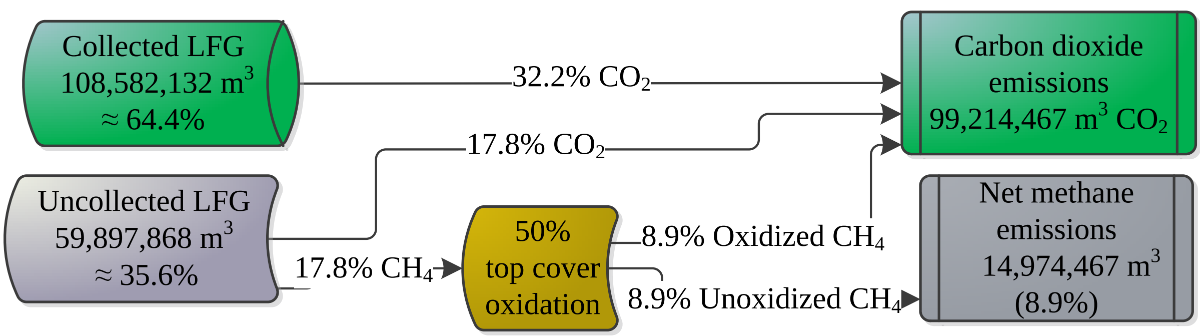


Figure 7-15: Volumes of annual carbon dioxide and net methane emissions for System option 3 for 10 years active operation time

In this system option the annual net methane emission and global warming contribution from this system is 14,974,467 m^3 (8.9% of the LFG potential). The carbon dioxide emission is 99,214,467 m^3 CO_2. The greenhouse gas of concern is the methane that can be emitted ($CH_{4\text{-emitted}}$) into the atmosphere which corresponds to 8.9% unoxidized methane (Figure 7-15).

The emitted methane ($CH_{4\text{-emitted}}$ = 14,794,467 m^3 CH_4) is equivalent to 10,652,016 kg CH_4 (density of methane 0.72 kg/m^3).

The global warming factor from emission of LFG (GWF) from this system option is estimated using the methane emission ($CH_{4\text{-emitted}}$) by multiplying with a GWP_{CH4} of 21 (i.e.1 kg CH_4 = 21 kg CO_2- eq.).

Accordingly, the global warming factor amounts to:

GWF_{LFG} = GWP_{CH4} * $CH_{4\text{-emitted}}$

=21 * 10,652,016 = 223,692,336 kg CO_2- eq. per year.

This option has the annual potential contribution of methane emission into the atmosphere of approximately 223,700 ton CO_2-eq per 1,000,000 tons MSW (0.2237 ton CO_2-eq per ton MSW). Upon capture and utilization of LFG for electricity generation and within the Cleaner Development Mechanism (CDM) framework the carbon credits can be claimed for the avoided methane emissions.

Sensitivity Analysis

A sensitivity analysis of this system is performed by altering some parameters and follow the changes in the performance of the system. The same general assumptions already made in the performed previous calculations such as the LFG collection efficiencies of the two reactor types, CH_4-CO_2 ratio, the methane oxidation by the top cover, the GWP_{CH4}, also apply in this sensitivity analysis. The following aspects were studied:

- Effect of the variation of years of active operation, varying between 10 and 200 years, on the annual amount of unconverted LFG;
- Effect of the length of operation time, varying between 5 and 15 years, on the annual landfill gas production, methane emissions and GWF;
- Effect of collection efficiency in LFB varied between 75% and 85%;
- Effect of change of the assumed 70 kg biogas produced from 300 kg OM present in the biowaste on the amount of biogas produced in the BIOCEL-system. Now the calculation included the assumption of biogas production of 70 kg biogas produced from 200 and 400 kg OM/ton wet waste (ten Brummeler 1993);
- Effect of 5 years operation time, 75% collection efficiency, 25% oxidation by top cover and 70 kg biogas produced in the BIOCEL-system from 200 kg OM/ton wet waste.

The first parameter we will discuss is the effect of the variation of the operation time of cells being active for a longer period than 10 years and still maintaining the 5 years of recirculation of leachate (cells are closed after 10 years). It is important to calculate the effect of active operation times different from 10 years. We look at the potential amount of gas that can and will escape into the atmosphere even with a 50% oxidation by the top cover. The baseline is the potential annual amount of LFG and methane still in the landfill site obtained from calculation is 42,863,000 m^3 LFG and 15,431 tons CH_4 respectively. For operation time varying from 10 years until 200 years then the amount and percentage of unconverted LFG potential and amount of fugitive gases are presented in Table 7-10.

Table 7-10: Annual unconverted (not produced) amount of potential LFG and potential CH_4 emission from the unconverted LFG in System option 3 as a function of active period of operation (varying from 10 to 200 years)

Years of operation	10	15	25	50	100	150	200
Unconverted LFG (x1000 m^3)	42,863	33,342	24,804	14,593	5,103	1,613	329
Potential CH_4 produced (tons)	15,431	12,003	8,930	5,253	1,837	581	118
Potential CH_4 emission (tons)	7,715	6,002	4,465	2,627	918	290	59

The tonnage and volume of gaseous potential emissions still left in the landfill site can be deduced from Figure 7-14. The amount of LFG still remaining at the site after closing is also 42,863,000 m^3. Until 25 years of operation, the methane potential is close to 5,000 tons per year. After 100 years none of the rapidly and moderately biodegradable organics will be left at the landfill site and the potential methane emissions are below 1000 tons per year as shown in Figure 7-16.

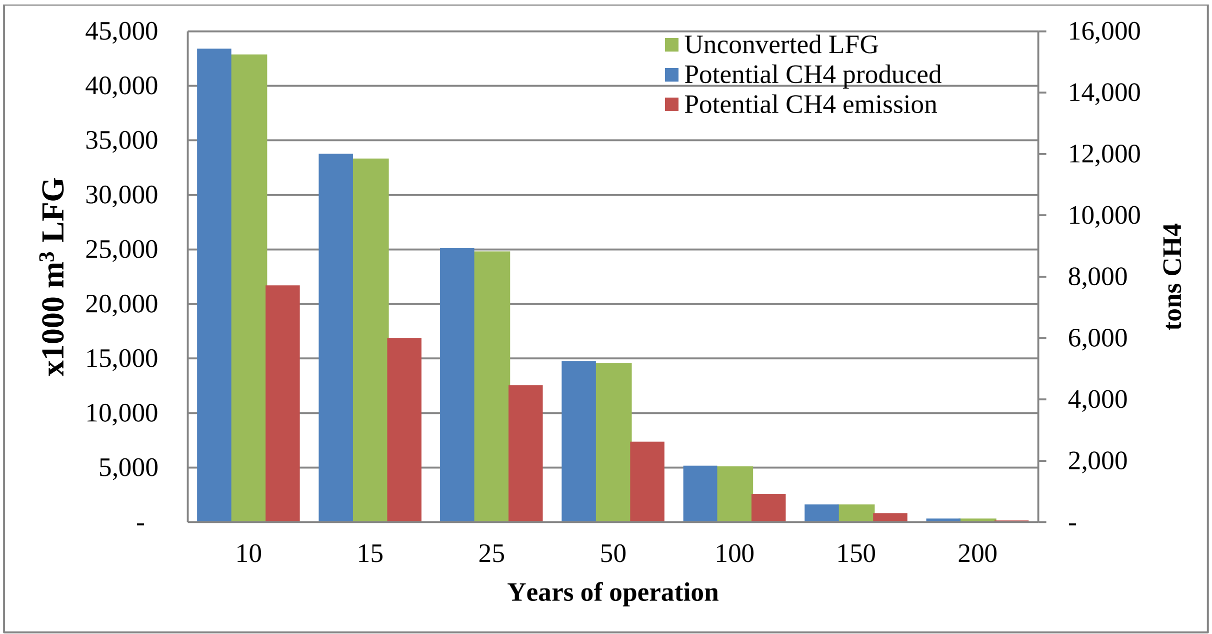


Figure 7-16: Annual unconverted LFG and potential CH_4 emission to the atmosphere by System option 3 as a function of the active period of operation (varying from 10-200 years)

Then we carried out the sensitivity of by changing the length of operational time. Table 7-11 is a summary showing the variation in LFG production and collection (80% collection efficiency) and the amounts of methane and carbon dioxide equivalent that can be emitted into the atmosphere versus 5, 10 and 15 years of active operation time to produce the potential 168,480,000 m^3 LFG.

Table 7-11: Effect of operation time on the annual landfill gas production and collection and the annual emission of methane and for System option 3 (Collection efficiency: 80%)

Operation time (years)	**LFG production** (m^3LFG)	**LFG produced** (%)	**LFG collected** (m^3LFG)	**Emitted** (m^3 CH_4)	**GWF** (ton CO_2- eq.)
5	103,631,583	61.5	90,994,155	19,371,461	292,896
10	125,616,554	74.6	108,582,132	14,974,467	223,692
15	135,137,922	80.2	116,199,226	13,070,193	197,621

A Standard LFB coupled with a BIOCEL system with 5 years operational time is capable of producing 61.5% of the LFG potential at a rate of about 103,632,000 m^3 LFG per year. Using an additional 5 years, almost 75% of all O.M present in the reactor will be converted to LFG. If the operation time is 15 years, the percentage LFG produced will increase again by 6% to about 80%. With respect to the GWF, a prolongation of the operation time from 5 to 15 years will produce a reduction from about 293,000 to about 198,000 ton CO_2- eq. This is a reduction of 33%. It is evident that the 10 years of operation are a more optimal time than a short 5 years or a relative long 15 years period to achieve 80% LFG production.

We also carried out a sensitivity analysis by changing the collection efficiency of the LFG production of 125,616,554 m^3 per year for the 10 years operational time (Table 7-11) and assessing the effect on the GWF. The results are presented in Table 7-12 and Figure 7-17.

Table 7-12: Effect of change of annual LFG collection efficiency on collected LFG, emitted amount of methane and GWF for System option 3 (Operational time: 10 years)

Efficiency (%)	**LFG collected** (m^3LFG)	**Emitted** (m^3 CH_4)	**GWF_{LFG}** (ton CO_2- eq.)
75	104,323,526	16,039,118	242,511
80	108,582,132	14,974,467	223,692
85	112,840,737	13,271,025	210,316

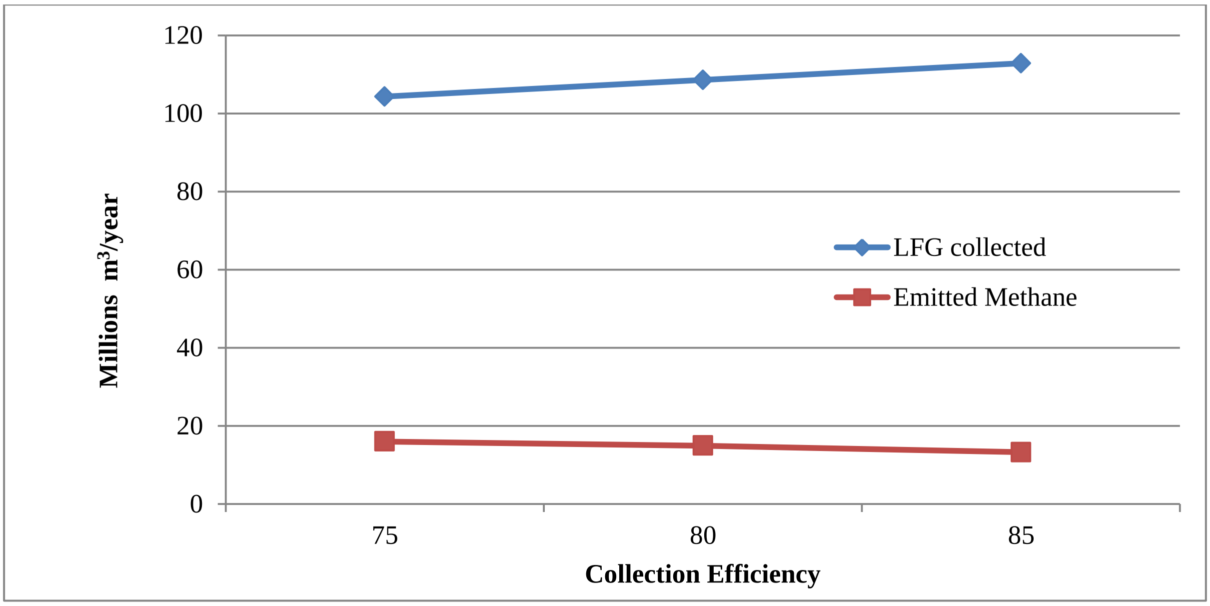


Figure 7-17: Effect of change of LFG collection efficiency on collected LFG and emitted methane for System option 3 (million m^3/year) (only a variation in collecting efficiency)

The influence of change of collection efficiency on the annually produced LFG is 4 million m^3 and a slightly significant difference is observed on the GWF whereby difference of 10% collection efficiency from 85% to 75% brings about 30,000 Tons CO_2-eq. per year but the drop in emitted methane is relatively small.

The fourth aspect we investigated is the effect of the assumed content of OM of the waste used for biogas calculation in the BIOCEL-system. The value assumed in section 7.4.3 is 70 kg biogas produced per 300 kg OM present in one ton of wet waste. In the sensitivity analysis we use the assumptions that 70 kg biogas is produced from 200 or 400 kg OM per ton of wet biowaste. The effect of the various assumptions is shown in Table 7-13 and Figure 7-18.

Table 7-13: Effect of the assumed OM per ton of biowaste in the BIOCEL system for the production of 70 kg biogas per ton of biowaste on annual LFG production, LFG collection, methane emission and GWF for System option 3

OM content (kg OM/ton waste)	**LFG production** (m^3LFG)	**LFG produced** (%)	**LFG collected** (m^3LFG)	**CH_4 Emitted** (m^3 CH_4)	**GWF_{LFG}** (ton CO_2- eq.)
200	126,507,528	75.1	113,339,356	13,785,161	208,432
300	125,616,554	74.6	108,582,132	14,974,467	223,692
400	125,421,436	74.4	106,403,816	15,519,046	234,648

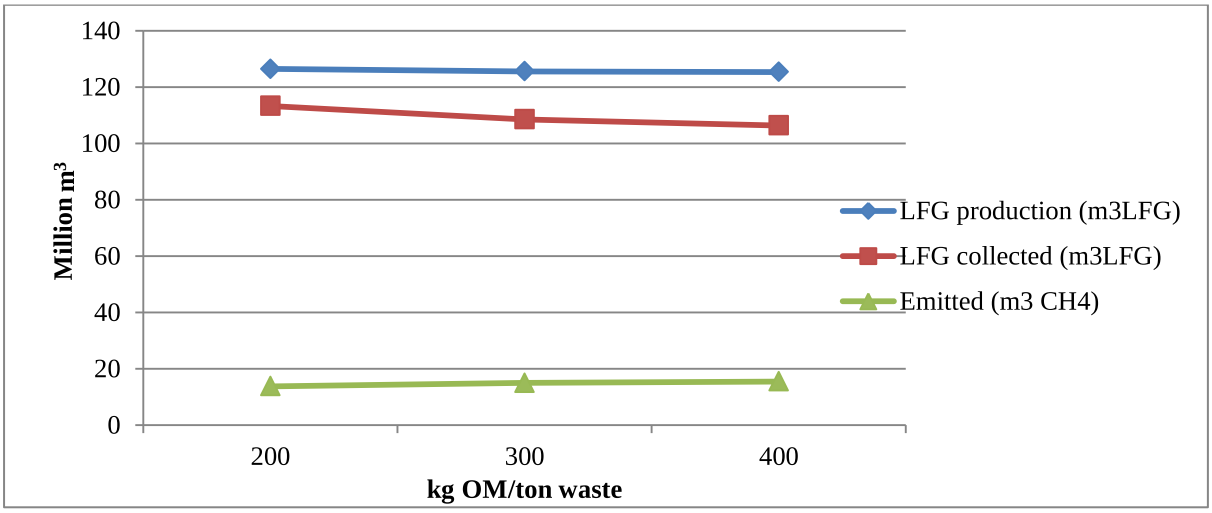


Figure 7-18: Effect of variation of OM per ton of biowaste in the BIOCEL on annual LFG production, collection and methane emission for System option 3

From Fig 7-18 the conclusion can be drawn that value of the biodegradable fraction in the waste has in particular an effect on the LFG collected and much less on the overall amount of LFG produced and the methane emitted. The higher the fraction of OM converted in the BIOCEL system the higher the total amount of LFG collected. This is mainly due to the 100% collection efficiency of the BIOCEL.

Finally we compared two situations both at 5 years of active operation but different LFG collection efficiencies, oxidation efficiencies and the fraction of organic matter converted to biogas. The situations are follows:

- ***Situation 1:*** 5 years of active operation, 80% collection efficiency of LFG, 50% oxidation efficiency by top cover and conditions of the BIOCEL-system corresponding with a production of 70 kg biogas from 300 kg biodegradable organics;
- ***Situation 2:*** 5 years of active operation, 75% collection efficiency of LFG, 25% oxidation efficiency by top cover and conditions of the BIOCEL-system producing 70 kg biogas from 200 kg organics.

Table 7-14 shows the results of the two situations with different LFG collection efficiencies, oxidation efficiencies and the fraction of organic matter converted to biogas.

Table 7-14: Annual amount of produced and collected LFG, methane emissions and the corresponding GWF for Situation 1 and 2 of System option 3

Situation	**LFG production** (m^3LFG)	**LFG produced** (%)	**LFG collected** (m^3LFG)	**Emitted** (m^3 CH_4)	**GWF_{LFG}** (ton CO_2- eq.)
1	103,631,583	61.5	90,994,155	19,371,461	292,896
2	107,276,224	63.7	95,623,834	9,107,021	137,698

Situation 2 shows more than 50% reduction of methane emissions and consequently of the GWF as compared to Situation 1. If applied in situations where the active operation time of the landfill site is relatively short and the oxidation efficiency of methane in the top layer is relatively low, the application of the BIOCEL-system is very beneficial in particular with respect to the greenhouse gas emissions.

7.4.6 Leachate production and recirculation

Leachate production in this option is the same or slightly more than that of System option 1. Part of the leachate is produced during the 20 days of biodegradation in the BIOCEL-system. From section 7.4.3 can be inferred that the amount of water still in the waste after biodegradation in the BIOCEL-system is 524,000 tons/year. Therefore the amount of leachate produced by the BIOCEL reactors is 640,000 - 524,000 = 116,000 tons/year. Another part of the leachate is produced during biodegradation of the waste in the LFB. As is the case for System option 1 and 2, leachate is collected and recirculated in both the LFB and the BIOCEL reactors. Recirculated leachate discharged from the BIOCEL-system is mixed with leachate emanating from the LFB. Figure 7-19 is an illustration of the water balance between the LFB, the BIOCEL, the leachate storage and recirculation and the leachate treatment plant. The quantity of leachate for recirculation in the LFB is the sum of the leachate generated from the LFB (Q_f) and that from the BIOCEL (Q_k). Excess leachate (Q_j) is diverted to a leachate treatment plant (LTP) for post-treatment and the effluent (Q_d) is discharged into receiving water bodies or municipal sewers (Note that here Q_j.= Q_d). When need arises for additional water to be recirculated into the bioreactor, then treated leachate from the LTP can be used.

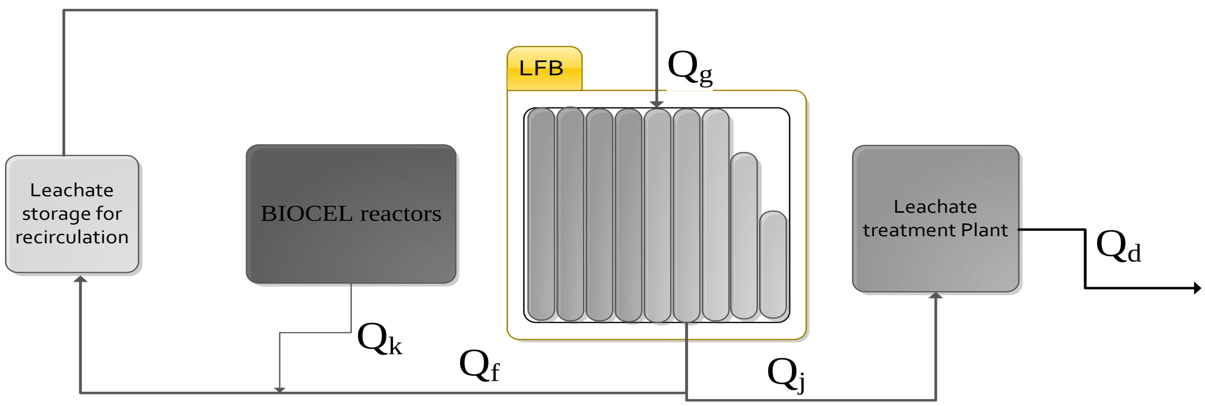


Figure 7-19: Water balance for System option 3

7.4.7 Leachate treatment

Leachate treatment for this system option is the same as that of System option 1. The treatment options are as discussed in chapter 4 geared towards removing mainly COD and nutrients, particularly Nitrogen- Ammonia.

7.5 System option 4: TWO STAGE TREATMENT – STANDARD LFB COUPLED WITH DECENTRALIZED BIOCEL AT TRANSFER STATIONS

7.5.1 General layout of option 4

This system option is a two stage treatment as illustrated in Figure 7-20 whereby waste is first pre-treated biologically for 20 days by means of decentralized BIOCEL reactors located at the MR-TF sites. It is assumed that the water content (water fraction) of the waste leaving the BIOCEL-system is the same as the water content of the waste entering the BIOCEL-system. Excess leachate is then transported to the landfill site and added to the leachate recirculation system at the LFB.

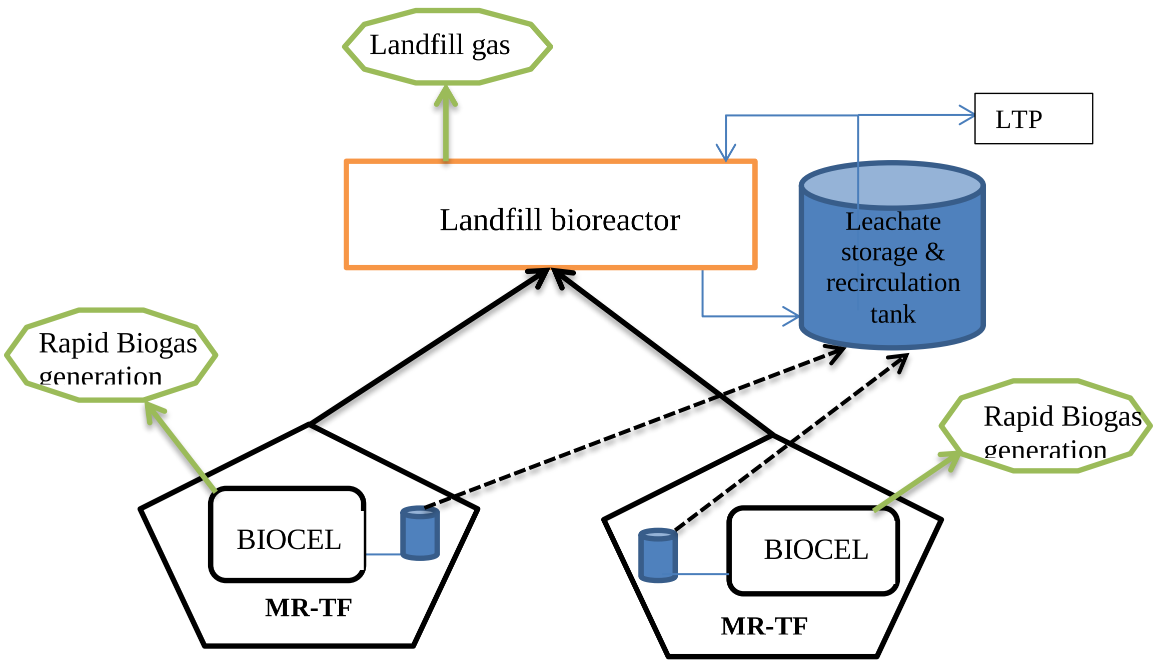


Figure 7-20: System option 4 Two stage treatment – Decentralized BIOCELs and LFB

The waste leaving the BIOCEL-systems is disposed in the cells of the LFB. LFG is generated and collected and leachate is recirculated and excess leachate is treated in the LTP. The LFB and decentralized BIOCEL reactors proposed here are operationally the same as those in System option 3.

The strength of this option is reduction of the amount of waste to be transported and disposed of at the LFB, so that in the long run a cost reduction can be realized as less land space is needed. Another strength of this option is the possibility of direct use of the generated gas as a utility at the MR-TF. However, its weakness is the higher investment and operation and maintenance costs for the decentralized BIOCEL reactors at the MR-TF and the need for transportation to the LFB site of excess leachate which is not needed for recirculation in the BIOCEL reactors at the MR-TF.

7.5.2 Materials Recovery and Transfer station Facility

The MR-TF proposed for this option not only serves the purpose of sorting the biodegradables from the recyclables and reusables like in all other system options but also provides anaerobic pre-treatment of the waste in a BIOCEL-system. Therefore every MR-TF has its own BIOCEL-system. The characteristics and composition of waste reaching the MR-TF is similar to that of the previous options as shown in Table 7-1.

MSW collected from waste generation points is dealt with similarly to System option 1 as discussed in section 7.2.2.

7.5.3 BIOCEL-system

The operation of the decentralized BIOCEL-system is exactly the same as that proposed for System option 3. The difference between System option 3 and 4 is the size whereby these decentralized BIOCELs are smaller to cater for a only the amount of waste reaching the transfer station located at the MR-TF. Before treatment the fresh waste is seeded with processed waste. Each of the ten projected transfer stations with a decentralized BIOCEL-system is proposed to have a capacity of at least 332.5 tons/day (i.e. 940,000 tons/year plus one third of the waste as seeding material filled in 10 stations during 365 days). At the same 4 metre waste working height and a reactor footprint of 29 by 29 sq. metres , similar to that of Option 3, at least four such reactors should be available to pre-treat waste with a density of 0.5 ton/m^3. The land requirement for the reactors alone is 29 * 29 * 4 = 3,364 m^2 = 0.3364 ha.

The amount of waste to be transported and disposed at the landfill site after the 20 days biodegradation in the BIOCEL-system is a function of how much solids, particularly organics and water, this system removes as described under System option 3. Accordingly, the total amount of wet waste leaving all decentralized BIOCEL reactors after 20 days of degradation is 770,000 tons wet waste/year. This is the amount of waste to be transported to the LFB.

7.5.4 Landfill Bioreactor

In System option 4, the LFB is similar to the one described in section 7.2.3, where 770,000 tons wet waste/year is received from the BIOCEL-system. For the same cell filled with 10 m high waste at a density of 0.5 ton/m^3 the total surface area required is 154,000 m^2/year equivalent to 15.4 ha per year. All operations and processes of opening, filling and closing of cells, leachate collection and recirculation systems as proposed for System option 3 are also applied for this option.

7.5.5 LFG generation in System option 4

Landfill gas generation in this option occurs in two separate reactor types similar to System option 3. LFG is first generated at the decentralized BIOCEL reactors located at the MR-TFs and subsequently in the Standard LFB at the disposal site. The gas composition is 50% CH_4 and 50% CO_2 v/v at STP. The LFG potential is the same as that of System option 1 which is 168,480,000 m^3 LFG. The major difference with System option 3 is the biogas collection from the decentralized BIOCEL reactors. Assuming all BIOCEL-systems will produce an equal amount of biogas, each individual reactor will potentially produce 4,044,444 m^3 and the remaining LFG is produced at the LFB.

Estimation of greenhouse gas reduction

The overall estimation of greenhouse gas reduction associated with this system option is typically similar to that of System option 3 presented in section 7.4.5. As already mentioned in section 7.2.4, methane directly from the LFB is considered to be a GHG, but carbon dioxide is not. Carbon dioxide emissions directly from the waste mass are a biogenic by-product and as such not included in the global warming factor.

Estimation of greenhouse gas emission associated with the decentralized BIOCEL-system and Standard LFB is made based on the following:

- No greenhouse gas emission from the decentralized BIOCEL-system because collection is 100%;
- Small amount of greenhouse gas emission during 10 years of operation due to uncollected LFG (practical LFG collection efficiency at the LFB is 80%);
- During and after 10 years of operation the typical oxidation efficiency by the LFB top cover is 50% of the methane that passes through.

The annual net methane emission and global warming contribution from this system is similar to that of System option 3 which is 14,974,467 m^3 (8.9% of the LFG potential) and carbon dioxide emission is 99,214,467 m^3 CO_2. The greenhouse gas of concern is the methane that can be emitted ($CH_{4\text{-emitted}}$) into the atmosphere which is the 8.9% unoxidized methane (Figure 7-16).

The emitted methane ($CH_{4\text{-emitted}}$ = 14,794,467 m^3 CH_4 is equivalent to 10,652,016 kg CH_4 (density of methane 0.72 kg/m^3).

The global warming factor from emission of LFG (GWF_{LFG}) from this system option is equivalent to that of System option 3 which amounts to 223,692,336 kg CO_2- eq. per year (Table 7-12).

This option has an annual potential contribution of methane emission into the atmosphere of approximately 223,700 ton CO_2-eq. per 1,000,000 tons MSW (0.2237 tons CO_2-eq per ton MSW) like System option 3. Upon capture and utilization of LFG for electricity generation and within the Cleaner Development Mechanism (CDM) framework the carbon credits can be claimed for the avoided methane emissions.

7.5.6 Leachate production and recirculation

The excess leachate produced in this system option is transported to the landfill site where it is recirculated together with the LFB leachate. As is the case for System option 3, the water balance which defines the amounts of leachate storage and recirculation is the same. Thus reference is made to section 7.4.6 and as illustrated by Figure 7-19.

7.5.7 Leachate treatment

Leachate treatment for this option is the same as for System option 3. The treatment options are as discussed in chapter 4 and are geared to removing mainly COD and nutrients, particularly Nitrogen- Ammonia.

7.6 Comparison of the system options

This section presents a comparison of the proposed System options and a Controlled dump-site (existing situation in East Africa).

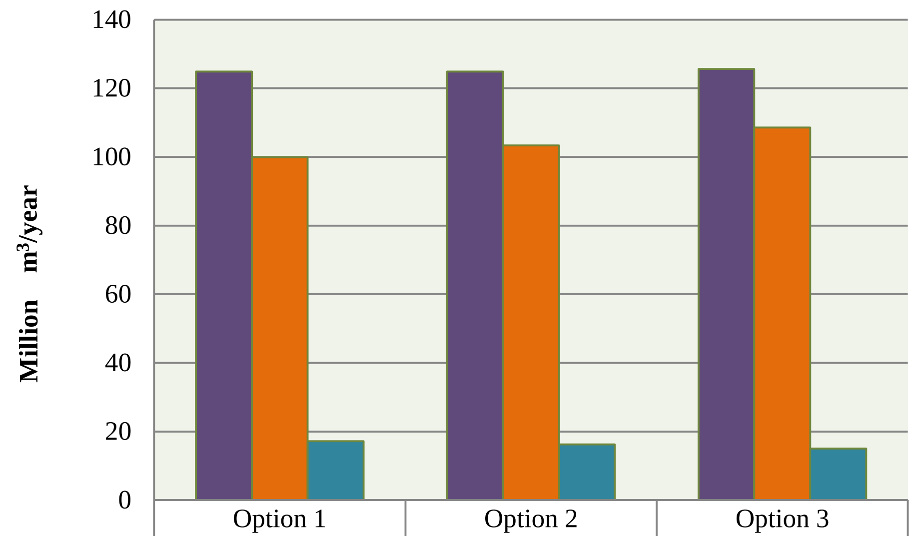
System option 4 is basically the same as System option 3. The difference is that System option 3 is a more centralized system whereas System option 4 has decentralized BIOCEL reactors so that the annual amounts of LFG produced and collected are basically the same and the potential methane emissions are also the same. As System option 4 gives the same outcomes as System 3 only System options 1, 2 and 3 and the Controlled dump are involved in the comparison. We have adopted a standard condition of 10 years of active operation of the landfill site, a collection efficiency of LFG in the LFB oft 80%, and gas collection efficiencies in the UASB and BIOCEL-system of 100%. The BIOCEL-system produces 70 kg biogas from 300 kg OM and the methane oxidation by the top cover of the site amounts to 50%.

7.6.1 LFG production, collection and methane emissions

Under the standard conditions mentioned above, all the system options generate almost the same amount of LFG in a 10 years active operation time which is about 125 million m^3 (≈74% of the LFG potential). System option 3 has a slightly higher LFG production (800,000 m^3/year) owing to the inclusion of the BIOCEL-system. The amount of LFG collected in System option 3 is 5 million m^3 higher than in System option 2 and 9 million m^3 higher than in system option 1. As a consequence of better LFG collection, System option 3 potentially emits 1.3 million m^3 and 2.1 million m^3 methane less than System options 2 and 1 respectively. All the system options have a significant methane emissions reduction as compared to the controlled dump. Table 7-15 and Figure 7-21 enable the comparison of the options by presenting the annual amount of LFG produced and collected on a 10 year active operation basis and the potential methane emissions produced by each system option.

Table 7-15: LFG produced, collected and methane emissions comparison of the system options

System options	Option 1	Option 2	Option 3	Option 4
LFG produced (m^3)	124,836,084	124,836,084	125,616,554	125,616,554
LFG collected (m^3)	99,868,867	103,370,470	108,582,132	108,582,132
CH_4 emitted (m^3)	17,152,783	16,277,383	14,974,467	14,974,467



	Option 1	Option 2	Option 3
LFG produced (m3)	124,836,084	124,836,084	125,616,554
LFG collected (m3)	99,868,867	103,370,470	108,582,132
CH4 emitted (m3 CH4)	17,152,783	16,277,383	14,974,467

Figure 7-21: System options comparison on annual LFG produced and collected and potential methane emissions

7.6.2 Optimization of LFG production

One of the ways to assess the systems is to set a certain target. For example for the optimization of the system options we set the target of collecting 80% of the 125,000,000 m^3 LFG produced which is 100 million m^3 LFG per year. A variation of the number of active operation years or the collection efficiency were the chosen values for the optimization. Table 7-16 shows the respective values of collection efficiency and years of operation for which the system options will achieve the set target.

Table 7-16: Optimization of the system options at a set target of 100 million m^3 LFG collected

Value	**System options**		
	Option 1	**Option 2**	**Option 3**
10 years Operation time	*Required collection efficiency*		
	80%	73%	70%
80% collection efficiency	*Required years of operation*		
	10 years	8 years	7 years

If operating actively during 10 years System option 3 will meet the set target at a collection efficiency of 70% while for system option 2 it will require an additional 3% collection efficiency to achieve the same. Under the same conditions System option 1 requires a collection efficiency of 80%. However, the lower the collection efficiency the higher the amount of uncollected LFG which in turn leads to more methane emissions to the atmosphere.

It would take 7 years for System option 3 to reach the LFG production target of 100 million m^3, whereas it would take 8 and 10 years respectively for System option 2 and 1 to reach the same target. The number of years of operation has an effect on the number of years for cells to be fully or partially active before closure and that has an effect on the overall size of the working area of the landfill site.

7.6.3 Comparison of methane emission between the system options and controlled dump

We assume that the controlled dump receives the same amount and same type of waste as the four System options and that the LFG potential of the controlled dump and the system options is the same, i.e. 168,480,000 m^3 LFG. In controlled dumps there is no LFG collection but only capping of the waste by a top cover. If for the controlled dump the same assumption were made about the oxidation of methane in the top cover as for the four System options, then the net methane emission from the controlled dump is 0.25*168,480,000 = 42,000,000 m^3 CH_4 (635,040 tons CO_2-eq. /year). Comparing the net methane emissions from the controlled dump and the four proposed System options, the System options provide a significant reduction in methane emissions to the atmosphere and thus GHG emission reduction. System options 3 and 4 reduce CH_4 emission three times (≈65% less) while System options 1 and 2 reduce by almost 60% as compared to the controlled dump.

Table 7-17: Comparison of CH4 emission between the system options and controlled dump

System option	**Net CH_4 emission** (m^3 CH_4)
1	17,152,783
2	16,277,383
3	14,974,467
4	14,974,467
Controlled dump	42,000,000

7.6.4 Alternative water content management in the waste

In the four elaborated system options the general assumption is that the wet waste including water in excess of the field capacity goes from the MR-TF to the BIOCEL-reactors or disposed in the cells of a LFB. However, it is also possible to have alternative leachate management which can have a positive impact on the size of the LFB and on the BIOCEL-system proposed in the four system options. The alternative is to drain the waste at the MR-TF and to remove the drained amount of water before the waste is transported to the disposal site. The amount of water that can be drained depends on the drainage system that is applied and the time available for drainage. A crucial factor is the field capacity (FC). As has been elaborated in the previous sections, it can be expected that at conditions of the disposal site the FC value is about 0.4. It is unlikely that we can achieve this value at the MR-TF. First of all the time available for drainage should be kept short in order to avoid serious odour problems. Nevertheless, because of the high water content of the waste at the MR-TF we might expect that a substantial part of the water present in the waste can be drained in a short period. In order to show the effect of a drainage step we assume that all water in excess of 1 kg water/1 kg dry solids (moisture content 50%) can easily be removed as drainage water.

In this section we discuss for each system option, the amount of water that can be drained and the consequences of the alternative draining of water before waste deposition.

System Option 1

The amount of wet waste at the MF-TF station after removal of 60 kg valuable products per ton is 940,000 tons/year. From that waste, 640,000 tons is water and 300,000 tons is solids (see table 7.1) and the water content is above field capacity. In the drainage process water is removed until the residual water content of the waste corresponds with 1 kg water /kg of dry solids. It means that the total amount of water in the waste at the end of the drainage process is 300,000 ton/year. The amount of water that is removed from the waste at the MR-TF is 640,000 – 300,000 = 340,000 tons water per year and therefore the remaining waste is 600,000 tons/year.

If the cells of the Standard LFB are filled with 600,000 tons/year of wet waste and using the same 10 m waste height and density of 0.5 ton/m^3 used in all the previous calculations, the required land space for the LFB is 120,000 m^2/year equivalent to 12 ha/year which is 6.8 ha/year less compared with the previous calculation (18.8 ha) in which all the wet waste was taken to the landfill site. The 340,000 tons water per year removed from the waste at the MR-TF can either be discharged into municipal sewers because it is expected that this water is only moderately polluted as biodegradation has hardly begun, or can be treated at the MR-TF in a separate treatment installation.

In the cells at the LFB site biodegradation takes place resulting in the conversion of organics into LFG at an average rate of 0.72 m^3/kg OM. In 10 years of active operation about 125,000,000 m^3 LFG is produced. This amount corresponds with an amount of organic matter ≈173,600 tons dry matter converted to LFG, It is assumed that after 10 years the waste in the site is at field capacity. After 10 years the total amount of waste left in the cells is 300,000 - 173,600 ton of waste = 126 400 ton of waste. The amount of water that is at field capacity is 0.4* 126,400 =50,560 tons. The total amount of water that is released from this waste is equal to 300,000 - 50,560 = 249,440 tons water which has to be treated at the leachate treatment plant at the landfill site. However, if all the leachate produced in this option would have to be treated by the leachate treatment plant at the landfill site then the amount would be 340,000 + 249,440 = 589,440 tons/year water.

For this system option there are three possibilities of how the removed water at the MR-TF can be handled:

- The 340,000 tons /year removed water at the MR-TF can be treated at the MR-TF in a separate treatment installation;
- The removed water at the MR-TF can be discharged into municipal sewers, This does not have to present problems as biodegradation has hardly begun and the water is only moderately polluted;
- The 340,000 tons/year removed water can be transported separately from the transport of the waste to the disposal site and treated at the landfill site.

Consequences of the modification of System option 1

- The eventual amount of waste to be transported from the MR-TF to the landfill site is 600,000 tons/year which is 340,000 tons/year less than the initial amount of waste resulting in less transport costs, less amount (volume) to deposit and thus less land space required;
- At the MR-TF a drainage system is required that brings the water content in the waste to a concentration of 1 kg water/kg dry solids;

- If the water removed at the MR-TF is not transported to the disposal site then a treatment or discharge system is required for 340,000 tons water per year at the MR-TF;
- If the water removed at the MR-TF is not transported to the landfill site, at that site a treatment system for 249,440 tons/year of leachate generated as a result of biodegradation in the LFB is also necessary.

System option 2

The alternative of water content management for this option is identical to that of System option 1. Cells of the Standard LFB coupled with a UASB reactor are also loaded with 600,000 tons/year of wet waste and using the same 10 m waste height used in all the previous calculations. In this case the required land space is 6.8 ha/year less compared with the original calculation in which all the wet waste was taken to the landfill site (18.8 ha). The annual amount of water that is removed from the waste at the MR-TF by drainage which is above field capacity is 300,000 tons and 249,440 tons is released at the landfill site as a result of biodegradation. The three possibilities of how the removed water can be handled mentioned under System option 1 also apply for this option.

Consequences of the modification of System option 2

The consequences of pre-drainage of leachate at the MR-TF for System option 2 are exactly the same as for System option 1.

System option 3

The alternative management of water content for System option 3 implies drainage of the water in the waste to a concentration of 1 kg water/kg dry solids at the MR-TF before the waste is taken to the BIOCEL-system at the disposal site.

In this case some more water is removed after 20 days of biodegradation in the BIOCEL-system. We assume that the water content of the waste leaving the BIOCEL-system is 1 kg water/kg dry solids and this waste is then deposited in the cells of the Standard LFB.

As it is the case for System option 1, the BIOCEL-system receives 600,000 tons/year of wet waste (with a water content of 1 kg/kg dry solids) and the water that is removed from the waste at the MR-TF is 340,000 tons water per year. Seeding material (also with a water content of 1 kg water /kg dry solids) is one third of the fresh waste. Consequently, the amount of waste to be filled in the BIOCEL reactors is 600,000 + 200,000 = 800,000 tons/year (≈2200 tons/day). Accordingly, 40 BIOCEL reactors are required that operate with 4 metre waste working height. The land space required for each BIOCEL reactor will be 24 by 23 = 552 m^2 and for 40 such reactors that adds up to 22,080 m^2 or 2.208 ha. This is 11,560 m^2 or 1.156 ha less land space required compared with the previous calculation (3.364 ha) in which all the wet waste was taken to the BIOCEL-system at the landfill site.

Biodegradation in the BIOCEL-system reduces the amount of organics by a factor 70/300 = 0.233 (i.e. 70 kg biogas is produced from 300 kg biodegradable OM) under the assumption that 1 kg biogas produced is equal to 1 kg organics removed. The amount of organics remaining in the waste from the BIOCEL-system is 0.767 *234,000 tons/year which amounts to ≈ 180,000 tons organics/year. The inert organics and inorganics (66,000 tons/year) remain unchanged so the total solids remaining in the wet waste leaving the BIOCEL-system amount to 180,000 + 66,000 = 246,000 tons per year. Because the water content of the waste leaving the BIOCEL-system is 1 kg water /kg dry solids the water content (on an annual basis) of this waste is also 246,000 tons. Therefore the total amount of wet waste that leaves the BIOCEL-system and is transported to the cells of the LFB is 246,000 tons solids + 246,000 tons water = 492,000 ton per year. For the same cell size as that of System option 1, filled with 10 m high waste at a density of 0.5 ton/m^3 the total surface area required is 98,400 m^2/year equivalent to 9.84 ha per year.

In the cells at the LFB site biodegradation takes place resulting in the conversion of organics into LFG at an average rate of 0.72 m^3/kg OM. In 10 years of active operation about 85,172,000 m^3 LFG is produced. This amount corresponds with an amount of converted organic matter of 118,300 tons. After 10 years the amount of water in the LFB is at field capacity meaning that the amount of water is 0.4 times the amount of solids. The amount of solids at the end of the active landfilling period is 246,000 - 118,300 = 127,700 ton per year. This means that the amount of water belonging to this amount of waste is 0.4 *127,700 ≈ 51,000 ton water/year. The total amount of leachate that is produced at the LFB site from the BIOCEL-system and from the LFB amounts to 300,000 - 51,000 = 249,000 ton water/year.

There are three possibilities of how the removed water can be handled:

- The 340,000 tons/year removed water at the MR-TF that can be treated at the MR-TF in a separate treatment installation;
- The water removed at the MR-TF can also be discharged into municipal sewers because it is only moderately polluted as biodegradation has hardly begun;
- The water removed at the MR-TF can be transported to the landfill site separately from the drained waste and be treated at the landfill site.

Consequences

- The amount of waste to be transported to the BIOCEL-system located at the landfill site is 600,000 tons/year which is 340,000 tons/year less than the initial 940,000 tons/year;
- With less waste reaching the BIOCEL-system transport costs are reduced, the amount (volume) to be deposited is lower and consequently 34% less land space is required for the BIOCEL reactors;
- At the MR-TF a drainage system is required that brings the water content to a value of 1 kg water /kg dry solids;
- If the water drained at the MR-TF is not transported to the landfill site, at the MR-TF a treatment system is required for 340,000 tons water/year;
- If the water drained at the MR-TF is not transported to the landfill site, at that site a leachate treatment facility is necessary for treating 249,000 tons leachate/year generated as a result of biodegradation in the BIOCEL-system and the LFB.

System option 4

The alternative management of water content for this system option is similar to that of System option 3 which involves draining of the water in the waste at the MR-TF before filling the waste in the decentralized BIOCEL reactors. The difference with System option 3 is that the BIOCEL reactors are located at the 10 MR-TFs thus with 4 BIOCEL reactors each. Another difference between this system option and System option 3 is that the leachate produced as a result of the 20 days of biodegradation in the decentralized BIOCEL-system remains at the MR-TF. The amount of biogas and leachate production is the same as in

System Option 3. Assuming that all MR-TFs receive equal amounts of waste, the amount of water that is removed from the waste at each MR-TF is 34,000 tons water per year and the remaining waste is 60,000 tons/year per MR-TF station (i.e. 600,000 tons/year from all 10 MR-TFs). The land space required for each BIOCEL reactor that operates with 4 metre waste working height will be 24 * 23 = 552 m^2. Four of these reactors require a land space of 2208 m^2 or 0.2208 ha which is less than if all the wet waste were treated by the BIOCEL-system at the landfill site which was (0.3364 ha).

The amount of water removed by drainage at each decentralized MR-TF is 34,000 tons per year and from the BIOCEL-system at the MR-TF 30,000 – 24,600 = 5,400 ton of water per year is released. Therefore at each MR-TF, the amount of leachate generated and requiring treatment is 34,000 + 5,400 = 39,400 tons of water per year.

The waste is then transported from all 10 MR-TFs to and deposited in cells at the LFB site. In the cells at the LFB site biodegradation takes place resulting in the conversion of organics into LFG at an average rate of 0.72 m^3/kg OM. In 10 years of active operation about 85,172,000 m^3 LFG is produced. This amount corresponds with an amount of converted organic matter of 118,300 tons. After 10 years the amount of water in the LFB is at field capacity meaning that the amount of water is 0.4 times the amount of solids. The amount of solids at the end of the active landfilling period is 246,000 - 118,300 = 127,700 ton per year. This means that the amount of water belonging to this amount of waste is 0.4 *127,700 ≈ 51,000 ton water/year. The amount of leachate that is produced at the LFB site amounts to 300,000 - 51,000= 249,000 ton water/year.

In System option 4 the possibilities of handling the removed water are similar to System option 3 with some additional possibilities. The possibilities are:

- The 39,400 tons/year removed water can be treated at each MR-TF in a separate treatment installation;
- The drained water before biodegradation in the BIOCEL-system can be discharged into municipal sewers because it is only moderately polluted as biodegradation has hardly begun;
- The water removed during biodegradation in the BIOCEL-system can be treated at the MR-TF site separated from the water drained before biodegradation in the BIOCEL-system;
- The water removed during biodegradation in the BIOCEL-system can be transported to the disposal site separately from the waste and be treated at the landfill site.

Consequences

- The amount of waste to be filled into the decentralized BIOCEL-systems located at the MR-TF is 600,000 tons/year which is less than the initial 940,000 tons/year;
- If waste is dewatered before being treated in the decentralized BIOCEL-systems, a lower amount (volume) has to be treated so that less space is required;
- If the water removed from the BIOCEL-system is treated at the MR-TF site, less leachate needs to be treated at the leachate treatment plant at the landfill site;
- At the MR-TF a drainage system is required that brings the water content to a value of 1 kg water /kg dry solids;
- At the MR-TFs a treatment or discharge system is required for the 39,400 tons/year water;
- At the landfill site a treatment system is necessary to treat 249,000 tons/year leachate.

7.7 Qualitative cost analysis of the four system options

This section provides a qualitative costs and benefits assessment in which the four system options are compared with existing disposal facilities (controlled dumps) in Tanzania, and in countries of East Africa, to enable a further evaluation of the proposed system options. The information presented here provides a framework for decision makers to evaluate the opportunities of the system options. The evaluation takes into consideration the necessity of having a sustainable waste management system based on the integration of technical, environmental, social and economic issues to achieve a healthy and liveable community long-term.

In a comparison with sanitary landfills studies have shown financial benefits of LFBs to include more efficient utilization of airspace, reduced leachate treatment/disposal costs, deferred new cell construction, earlier beneficial reuse of land, post-closure savings from fewer monitoring and financial assurance requirements, and more efficient gas collection resulting in larger revenues from energy production (Berge et al. 2009). These economic benefits may be diminished by costs associated with increased operating requirements, increased capital costs for leachate injection facilities and additional monitoring equipment to control LFB functions. Cost analysis studies executed by (Chong et al. 2005; Berge et al. 2009) have shown that LFBs are economically comparable, if not more advantageous, than conventional landfills.

Analysis to evaluate qualitatively the costs and benefits associated with LFBs is conducted in comparison with existing disposal facilities (controlled dumps) in Tanzania. A number of aspects is discussed in the following subsections. These aspects are:

- Investment and operation costs;
- Land space requirements for waste pre-treatment;
- Leachate treatment costs and savings;
- LFG generation and utilization and costs and benefits;
- Airspace recovery benefits;
- Greenhouse gas accounting and global warming avoidance.

7.7.1 Investment and operation costs

The investment cost components related to the construction of LFBs are listed in Table 7-18. They are basically construction and installation costs which in the order of project execution are the preparation of works, laying the bottom liner, installation of the leachate collection and recirculation system, installation of LFG collection systems and support facilities. The various main subcomponents under each stage are shown as well.

These basic cost components are related to the landfill size which are different for the various system options. The cost components depicted in Table 7-18 apply to all the proposed system options. Specific additional costs for the options 2, 3 and 4 in excess of the standard LFB (System option 1) are:

- System Option 2: construction of a UASB reactor and installation of gas collection pipes connected with the pipes of the LFB for collective gas accumulation;
- System Option 3 and 4: construction of BIOCEL-system reactors, installation of degassing and gas collection pipes also to be connected with the LFB gas collection pipes.

Table 7-18: Basic cost aspects associated with construction of a LFB system

Preparation works	Bottom liner	Leachate system	LFG collection systems	Gas utilization facilities	Support facilities
Land acquisition	Excavation works	Collection and recirculation pipes	Collection pipes	Gas cleaning devices	Management offices, facilities
Survey of the LFB site	Clay layer	Storage tank	Gas cleaning devices	Gas storage	Fencing
Clearing of the site	Geo-membrane	Leachate pumps and treatment plant	Appurtenances (valves, flow meters)	Energy generators and gas flares	Weigh bridge

Operation and maintenance costs include the expenses for support staff, running of facilities and equipment (filling the landfill site and reactors, leachate and gas treatment, fuel and electricity, costs for inspecting and repairing potential failure of the leachate plant and gas leakages and costs for additional staff for monitoring and engineering services.

7.7.2 Land space requirement for the system options

All the system options have some common land space requirements which are the need for MR-TFs and waste pre-treatment, landfill site, leachate storage and treatment site. The land space for MR-TFs, waste pre-treatment, leachate storage and leachate treatment are one-time costs with no recurrence, but the land space for the landfill site is based on an annual requirement. Table 7-19 shows a mixed quantitative and qualitative description of land space requirements for the proposed system options. The quantitative values are based on a waste input of 940,000 kg/year requiring disposal/treatment and an active operation time of cells of 10 years. The filling time of each cell is one week.

The annual land requirement for the LFB siting in System options 1 and 2 is the same (18.8 ha) while the requirement for the LFBs after the BIOCEL-system is 15.4 ha per year (System option 3 and 4) by virtue of conversion of a part of the organic material in the wastes. The space requirements for the MR-TFs for System options 1, 2 and 3 are the same (standard) but System option 4 needs an additional 0.3364 ha at each MR-TF (total number of MR-TF is 10) to be occupied by decentralized BIOCEL-reactors.

Table 7-19: Land space requirement for the system options

Component	System options			
	Option 1	Option 2	Option 3	Option 4
MR-TF	Standard	Standard	Standard	Additional (decentralized – BIOCEL at each MR-TF 0.3364 ha)
Waste pre-treatment	None	None	Additional (BIOCEL = 3.364 ha)	None
LFB site (ha/year)	18.8	18.8	15.4	15.4
Leachate storage and treatment	Standard	Additional (UASB reactor)	Standard	Standard

Like System option 4, System option 3 needs additionally 3.364 ha of land for the BIOCEL-system at the landfill site. In both options this space is needed only once at the start of the project. The inclusion of a BIOCEL-system whether centralized or decentralized (options 3 and 4) leads to a lower land requirement as the annual extra land needed for landfilling is less (15.4 ha) than in the system options 1 and 2 (18.8 ha).

7.7.3 Leachate treatment costs and savings

In controlled dumps, treatment of all leachate generated is compulsory. In general this is more costly than treatment in the four system options elaborated in this chapter. An advantage associated with LFBs is their capacity to pre-treat leachate in situ, with potential cost savings. Operating an LFB may result in considerable leachate volume reductions as compared to controlled dumps which in turn may lead to reduced leachate treatment costs. Furthermore, leachate being recirculated is less polluted compared to non-recirculated leachate. Amongst the four options, system option 2 (LFB coupled to a UASB reactor) is envisaged to have lower treatment costs because the leachate is treated in the UASB reactor whereby most or all the rapidly biodegradable organic matter is converted to biogas. The remaining options require further treatment of the leachate despite the in-situ treatment by the LFB.

7.7.4 LFG generation and utilization costs and benefits

The generation of electricity from captured LFG is a potential benefit associated with LFBs. In controlled dumps LFG is generated but not captured. In all the proposed system options once the bioreactor is capped, the bioreactor operation commences (i.e. leachate recirculation and gas collection), gas generation starts and the collected gas is further processed for utilization. Gas can be utilized in an internal combustion engine (most commonly used equipment) to produce electricity.

The total profits from LFG utilization are defined as the revenue from electricity generation minus the costs associated with the purchase and operation of the gas engine The latter costs are incurred over the entire gas extraction and utilization period. To determine revenues a basic market price of electricity (UNC/KWh) is used. In the current situation in Tanzania the tariff for general customers is 0.08 US$/kWh.

7.7.5 Airspace recovery benefits

Airspace recovery is the amount of space regained as a result of biodegradation of waste thus the volume reduction in a cell and in the landfill site in totality. The recoverable volume of airspace is be calculated based on the amount of settlement of waste achieved, landfill volume, and a reutilization factor. The reutilization factor is the fraction of the recovered volume that can generate revenue by additional waste placement.

In LFBs leachate recirculation enhances biodegradation of the waste so that volume reduction is faster than in controlled dumps. And with the incorporation of the BIOCEL-systems, there is even more landfill space saving due to reduction of the amount of waste to be landfilled. Actual airspace gains occur incrementally as waste degrades. Therefore, some monetary gain can be made continuously because additional waste can be placed as the space is regained. It is important to note that the additional waste placed in the landfill will result in increased substrate for LFG generation.

7.7.6 Greenhouse gas accounting and global warming avoidance

It is important to map GHG emissions from waste management. One key aspect in accounting GHG emissions is that most waste management technologies are sources of greenhouse gases, which can be reduced by minimizing LFG emissions. There are several reporting mechanisms for GHG emissions associated with waste management. In this thesis, a GWF is calculated from methane emissions to establish a carbon dioxide equivalent value per year. As already mentioned in subsections 7.2.4, 7.3.4, 7.4.5 and 7.5.5 of this chapter, carbon dioxide emissions directly from the waste mass is a biogenic by-product and as such is not included as part of the global warming potential.

In the existing system of controlled dumping no LFG is collected. The controlled dump has the same LFG potential as the four LFB system options but requires a longer time to degrade all wastes. In the controlled dump 50% of the potential methane gas generated is oxidized by the top cover; thus 25% of the biogas by volume is emitted as methane into the atmosphere. For a standard LFB (System option 1), standard LFB coupled with UASB reactor (System option 2), standard LFB coupled with BIOCEL-system (System option 3) and decentralized BIOCEL-systems and standard LFB (System option 4), all at 10 years active operation, it is envisaged that of the LFG generated respectively 74.1% for System option 1 and 2 and 74.6% for System option 3 and 4 can be collected and utilized for electricity generation. However, practically up to 80% of LFG that can be collected, so that the LFG collected by the system options 1 to 4 are 59.3%, 63.1%, 64.4% and 64.4% respectively. Table 7-20 presents the system options showing the percentage LFG produced and collected in 10 years of active operation and their respective emissions to the atmosphere. The percentages refer to the total LFG potential of the wastes 168,480,000 m^3. System option 3 and 4 exhibit more LFG collection than the other two options and thus less methane gas emissions by 2,100,000 m^3. System option 2 emits 1,600,000 m^3 CH_4 less than System option 1.

Table 7-20: Percentages of LFG produced and collected in a 10 years operational time and potential emissions from the four system options

System options	LFG produced (%)	LFG collected (%)	Emissions (m^3)
Option 1	74.1	59.3	17,152,783 m^3 CH_4 101,392,783 m^3 CO_2
Option 2	74.1	63.1	15,516,833 m^3 CH_4 99,756,833 m^3 CO_2
Option 3	74.6	64.4	14,974,467 m^3 CH_4 99,214,467 m^3 CO_2
Option 4	74.6	64.4	14,974,467 m^3 CH_4 99,214,467 m^3 CO_2

Table 7-21 is a summary of the global warming factors for the four system options excluding the carbon dioxide emissions which are biogenic so that they do not contribute to global warming and GHG accounting. Total emissions avoided by implementation of the system options is established as multiplication of the methane generated by the factor 21 as recommended by Intergovernmental Panel on Climate Change. Comparison is made between the system options and results show System option 3 and 4 have annual methane emissions lower by 11,000 and 36,000 Tons CO_2- eq. per year as compared to System option 2 and 1 respectively. The tonnage of carbon dioxide equivalence can be translated as reduction of 500,000 and 1,700,000 kg CH_4 by System option 3 and 4 than options 2 and 1 respectively.

Table 7-21: Potential methane emissions and GWF from the four system options

System options	Methane emitted (m^3 CH_4/year)	Methane emitted (kg CH_4/year)	GWF (Tons CO_2- eq. per year)
Option 1	17,152,783	12,350,003	259,350
Option 2	15,516,833	11,172,120	234,615
Option 3	14,974,467	10,652,016	223,692
Option 4	14,974,467	10,652,016	223,692

7.7.7 Summary of qualitative costs and benefits analysis

A summary of requirements, investment and O&M costs as well as potential monetary benefits from GHG emission avoidance for each system option is tabulated in Table 7-22. The costs are given in a comparison with a controlled dump which is the common current practice in most cities of East Africa. Costs that are the same for all options including the controlled dump are adopted as basic costs. The cost for waste pre-treatment by means of the MR-TF is an additional cost to the typical cost of a controlled dump which is applicable to System options 1 and 2. For System options 3 and 4, the costs of BIOCEL systems comes in addition to the costs of MR-TF whereby option 3 has a centralized BIOCEL system at the landfill site and system option 4 has decentralized BIOCELs located at the MR-TF. Then all LFB systems have additional costs for leachate collection, storage and recirculation systems, leachate treatment and LFG collection with an extra cost for System options 3 and 4 where more piping for both leachate and gas system will be required for the BIOCEL-system.

Savings are potential benefits that can be accrued from the use of these system options. The savings are in the reduced costs of leachate treatment in LFB systems because the leachate is partially treated by the recirculation back to the LFB. System option 2 realizes even lower costs because of the UASB used for rapid generation of biogas within the acidification regime as discussed in section 7.3.3. Another cost saving aspect is on the airspace recovery that can be achieved by operating a landfill as a bioreactor. This saving is the consequence of reduced acquisition of land for landfilling or extended life span of the landfill site on a long-term. Furthermore, the utilizable LFG can be a source of monetary benefits via the production of electricity and the CDM mechanism whereby carbon credits can be claimed in terms of the avoided emissions.

Table 7-22: Costs and benefits summary for the proposed innovative System options and Control dump as existing situation (as standard situation)

System option	**Investment and O&M Costs**				**Potential monetary benefits**			
	Land value, landfill preparation and operation	**Waste pre-treatment**	**Leachate storage and recirculation installation**	**LFG collection installation**	**Leachate treatment savings**	**Utilizable LFG generation (% of the potential amount)**	**Airspace recovery**	**GHG accounting/GWF**
Controlled dump	Basic costs **18.8 ha/year**	None	None	None	None Standard amount requiring full treatment	None	None or little amount after long period	25% net CH_4 emissions **Emitted 635,040** Ton CO_2-eq./yr.
Standard LFB Option 1	Basic costs **18.8 ha/year**	Additional (i.e. costs of the MR-TF)	Additional	Yes, on LFB	Less leachate (Polishing)	59.3%	Yes, costs saving is associated with the deferment of the next cell construction	10.2% net CH_4 emissions **Emitted 259,350** Ton CO_2-eq./yr.
LFB+UASB Option 2	Basic costs **18.8 ha/year** + Space for UASB	Additional (MR-TF)	Additional	As option 1 + additional for UASB	Less leachate	63.1%	Yes, costs saving is associated with the deferment of the next cell construction	9.2% net CH_4 emissions **Emitted 234,615** Ton CO_2-eq./yr.
BIOCEL+LFB Option 3	Basic costs **15.4 ha/year** + Space for BIOCEL	Additional + MR-TF + BIOCEL system costs	Additional	As option 1 + additional for BIOCEL-system	Less leachate Lower cost of leachate treatment	64.4%	Yes, costs saving is associated with less landfilled waste + Deferment of the next cell construction	8.9% net CH_4 emissions **Emitted 223,692** Ton CO_2-eq./yr.
Decentralized BIOCEL+LFB Option 4	Basic costs **15.4 ha/year** + Space for decentralized BIOCEL reactors	Additional + MR-TF + decentralized BIOCEL-system costs	Additional	As option 1 + additional for decentralized BIOCEL-system	Less leachate (Polishing)	64.4%	Yes, costs saving is associated with less landfilled waste + Deferment of the next cell construction	8.9% net CH_4 emissions **Emitted 223,692** Ton CO_2-eq./yr.

7.8 Conclusions

In this chapter we have developed and elaborated four innovative concepts of landfill bioreactor (LFB) system options and presented quantification of LFG production, emissions avoidance to the atmosphere and global warming contribution in comparison with the existing Controlled dumps. The four developed innovative concepts of landfill bioreactor (LFB) systems adaptable in East Africa are based on advanced existing knowledge and pilot scale experimental results. The developed innovative concepts comprise of materials recovery at transfer stations (MR-TF), large scale centralized LFB and other supporting reactors for waste degradation such as the BIOCEL-system. Introduction of the MR-TF is a crucial step in the improvement of municipal solid waste management in East Africa by moving away from the existing secondary collection points which only serve as storage and transfer points for the collected waste and for better performance of the proposed innovative options. From this chapter the following conclusions are drawn:

1. The four innovative system options for treatment of waste that have been elaborated for treatment of waste in a LFB are considered as technically feasible. These system options are:
    - System option 1: Standard Landfill bioreactor;
    - System option 2: Standard LFB with leachate acidification and LFG production in a UASB reactor;
    - System option 3: Two stage treatment – Centralized BIOCEL-system and LFB;
    - System option 4: Two stage treatment – Standard LFB coupled with decentralized BIOCEL-systems at MR-TFs.

2. By means of a semi- empirical model it is for a given set of input data possible to calculate the amount of LFG that is produced, the emission of methane, the size of the landfill site, and the amount of leachate that is produced and that has to be treated.

3. At standard conditions of the input parameters for the four system options which are:
    - Total annual amount of waste is1 000 000 ton;
    - Removal of 6% of the waste at the MR-TF station;
    - Composition of the residual amount of waste that has to be treated after removal of 6%: 681 kg water/ton, 249 kg biodegradable organics/ton, 71 kg inerts (organic and inorganic)/ton;
    - Density of the waste: 0.5 ton/m^3;
    - 10 years of active operation of the landfill site;
    - Collection efficiency of LFG: 80% in the LFB, 100% in the UASB and BIOCEL-system;
    - Production of biogas by means of the BIOCEL-system of 70 kg biogas from 300 kg biodegradable OM;
    - Methane oxidation in the top cover of 50%;
    - Composition of LFG (biogas) is 50% methane and 50% carbon dioxide;

The following results can be calculated; LFG potential, LFG production, greenhouse gas emissions, size of the site, amount of leachate to be treated.

4. At standard conditions the LFG potential from 1,000,000 collected waste and after removal of recyclables at the MR-TF leaving 940,000 tons/ year wet waste for disposal is

168,480,000 m^3/year. The annual LFG production based on 10 years of active operation by the system options is 125 million m^3 (74 -75% of the potential amount) for all system options. The greenhouse gas emissions (net methane emissions) by the proposed system options is less than half the emissions from the existing controlled dumps. Between the proposed options, the system options with BIOCEL-system have lower net methane emissions than the other two options. Greenhouse gas emissions from the system options are more are less identical being within the range of 224,000 – 260,000 tons CO_2 eq. per year.

5. At standard conditions of the input parameters the size of the site is 18.8 ha for System option 1 and 2. However, the size of the site for System options 3 and 4 is smaller than that of System options 1 and 2. For System option 3 it is 15.4 ha for the LFB and 3.364 ha for the centralized BIOCEL-system. For option 4 the size of the LFB is same as that of System option 3 (15.4 ha) and for the decentralized BIOCEL-systems the size of each BIOCEL-system at the MR-TF is 0.3364 ha under the assumption that there will be 10 of such systems.

6. The amount of leachate to be envisaged to be produced and requiring treatment is the same for System options 1,2 and 3 which is 589,354 tons/year. For System option 4 the amount of leachate envisaged to be produced is the same as that of system options 1, 2 and 3 but the amount of leachate that reaches the landfill site is less because of conversion of organics and concomitant removal of water from the wastes that has taken place at the decentralized BIOCEL-systems.

7. Active operation of 10 years is considered adequate for the Standard LFB because by that time at least 74% of the potential LFG is already produced which is a significant amount. Five years of active operation is too short mainly because by that time only 58% of organics are converted to LFG which also means only 58% of the potential LFG is produced. 15 years of active operation is too long as there are not much gains in terms of conversion of organics and LFG generation (only 6% additional compared to10 years of operation).

8. The amount of LFG collection and greenhouse gas emissions is not very sensitive for small changes in collection efficiency of the LFG, active operation time at the site and acidification regime of System option 2. However, acidification of the leachate in the new cells for System option 2 and conversion of volatile fatty acids to methane in a separate UASB reactor may require more management capacity but is particularly helpful when the recovery efficiency of the LFB is low.

9. If the gas collecting efficiency and /or the conversion of the biogas in the top cover are low, application of a BIOCEL anaerobic pre-treatment reactor has a strong favourable influence on the size of the site, and also on the emission of greenhouse gases.

10. More accurate parameters are required to make better and more reliable calculations particularly on the biogas potential of the BIOCEL-system, oxidation efficiency of the top cover of landfill bioreactors, recovery efficiency of the LFB, emissions of greenhouse gases and the length of time during which the acidification regime of the leachate can be maintained. This data can be obtained via experimental research.

11. All the four LFB system options cost more than the controlled dumps due to the addition of the leachate recirculation system and the gas collection system. System options 3 and 4 with the BIOCEL-system are even more costly than the other options. In general, benefits can be gained by electricity production and by claiming the CDM credits from the net avoided emissions. These gains could partly offset the costs of the systems.

12. The developed calculation model can easily be applied with input parameters which are different from the standard input parameters. Together with the four system options the calculation model provides a useful tool for decision making regarding municipal solid waste management in East African countries.

CHAPTER 8

Discussion and Conclusion

8.1 Introduction

Anthropogenic activities create waste, and it is the way these wastes are collected, stored, transported and disposed of, which pose risks to the environment and to public health. In developing countries especially municipal solid waste (MSW) causes a serious problem. An evaluation of the MSW management practice in East Africa found out that the major problems associated with MSW management in these developing countries include collection, transportation and disposal. Often waste collection systems are far from covering all communities and the waste that is collected is not treated in an environmentally sound manner. In many East African cities collected waste is simply deposited at dumpsites. This causes serious soil, groundwater and air pollution and health impairment and neglects possibilities for resource recovery, re-use and recycling.

A safe and reliable long-term disposal of the collected MSW is an important component of integrated sustainable waste management. So there is a strong need to find a sustainable solution that fits the local conditions of East Africa technically, economically, environmentally, societally and internationally where earnings can be gained by lowering the greenhouse gas emissions and collection and final use of biogas. Environmental concerns that date more than a decade ago predicted that, future waste management plans will include resource-conservation and source separation programs to enhance resource-recycling to re-process wastes into useful products, incineration to reduce the volume of waste and to recover energy from waste fractions that cannot be reused economically and new landfill design and operation technologies to dispose wastes in an environmentally sound manner and to recover energy. However, waste treatment technologies such as incineration, aerobic or anaerobic digestion systems as stand-alone systems are not feasible options for East Africa for the next 10 years. It is expected that more advanced landfills technologies are the most feasible options to solve the problem of MSW in the next 10 to 15 years.

This thesis addresses the need for East African cities of a cost-effective, land-saving and energy producing waste treatment technology, based on a sophisticated landfilling of the waste. Among the various sanitary landfill options the (anaerobic) landfill bioreactor (LFB) has been selected as a most promising technology, either as stand-alone system or in combination with certain pre-treatment technologies. Accordingly the main objective of this thesis is to have developed and described landfill bioreactor based municipal solid waste treatment systems suitable for East African cities. First an inventory and collection of relevant knowledge on LFB was gathered and synthesized. An experimental research into the biological acidification of waste and the production of biogas was carried out. Also models for calculation of the biogas production have been evaluated. As a second step, four innovative system options of an LFB have been developed, elaborated and evaluated by means of a desk study, including the use of mathematical models to calculate the production of biogas mass balances and comparisons. Then a critical look back at the evaluation and elaboration of the system options is made and finally conclusions and recommendations are put forward.

8.2 Collection and assessment of information relevant to development of LFBs in East Africa

8.2.1 Composition of waste in East Africa

The first step in this thesis was an empirical study, experimental research and executions of some desk studies to collect basic information that could be useful for the set up and evaluation of the four innovative LFB system options.

An empirical research focused on diagnosis of municipal solid waste management in rapidly growing cities of East Africa a case study of Mwanza, Tanzania was picked to represent other cities of the region. From the diagnosis the activities carried out include the making of an inventory for the current practices in waste collection and disposal in the study area (Mwanza - a typical rapidly growing city in East Africa) and the characterization of the collected MSW. Knowledge of waste characteristics and composition is indispensable for an appropriate choice of systems to manage the waste in a particular locality and it is in particular needed for the elaboration of suitable innovative treatment options for Tanzania and East Africa. The diagnosis involved identification of problems through desk study; interviews and questionnaire administration; field observation of practices and participation in MSW management activities, solid waste sampling and characterization which was conducted from October 2007 to March 2008. This assessment looked at practices pertaining to generation, storage, collection, transportation and final disposal of wastes. An experimental work included waste sorting and analysis activities to facilitate direct measurement of waste volume and composition during both the dry and the rainy seasons. The average generation rate found during this study was 0.32 $\pm$ 0.06 kg/cap/day comparable with previous studies conducted in Dar es Salaam –Tanzania whereby the domestic waste generation rate established ranged from 0.3 to 0.7 kg/cap/day. Findings reveal that on a dry basis over 84% is organic in nature and 14% consists of potentially recoverable materials and 2% are other materials such as e-waste, batteries, ceramics as shown in Figure 8-1.

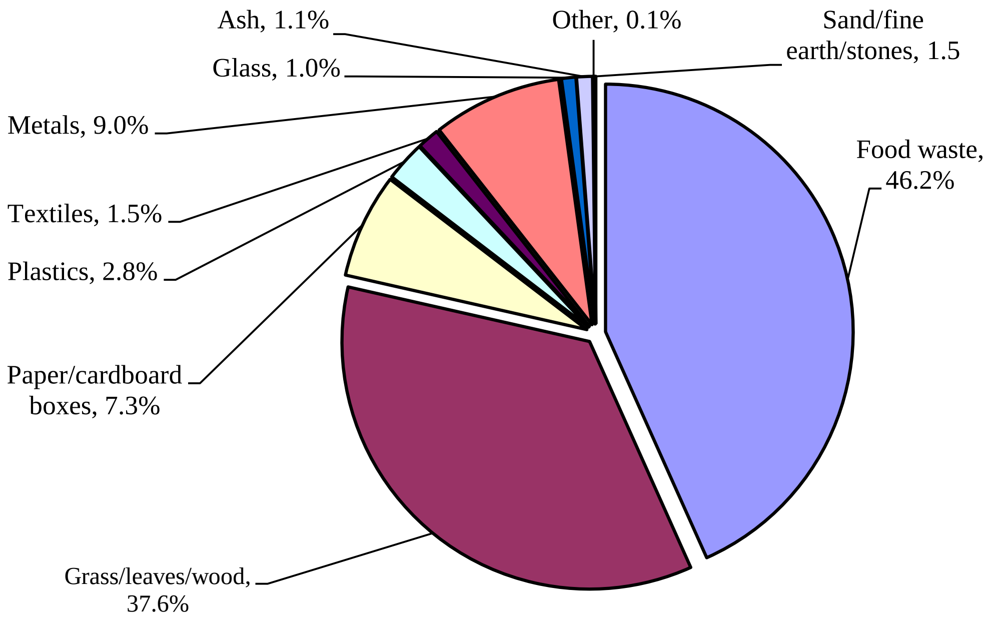


Figure 8-1: Percentage distribution of waste in typical city on East Africa

For calculation purposes we adopted that biodegradable organics in the waste is 65% (i.e. food waste and half of grass/leaves/wood) on dry basis and 35% as inert on a dry basis (i.e. 29% organic and 6% inorganic non-biodegradable). The moisture content of the waste amounts to 64%. This moisture content is extremely high if compared to the moisture content of MSW that is produced in highly industrialized Western countries.

Among the MSW management challenges found during the study, the key challenge was the extent of inappropriate waste disposal. The only current method of disposing of waste in East Africa is landfilling, in fact controlled dumping. Such dumpsites are commonly located on the out skirts the city centre and not completely fenced and most have no weigh bridge. Filling of waste is in cells but not clearly separated and not well planned thus the average height of the dumped waste is very low not more than 1 m leaving the waste scattered over a large area. This triggered the necessity to design and test viable MSW treatment options to suit the Tanzanian and East African context as a whole. Incremental improvements in landfill design and operation are more likely to succeed than attempts to make a single, large leap in engineering expectations. Thus the idea of operating the existing disposal techniques widely practiced i.e. landfill as a bioreactor at pilot scale was conceived.

8.2.2 State of the art and information relevant to LFB

A desk study on landfill bioreactors was conducted with the aim to generate insights to make well founded choices about the introduction of anaerobic landfill bioreactor technology and its applicability in East Africa. MSW placed in a landfill undergoes a number of simultaneous and interrelated biological, chemical and physical processes related to the conversion of the organic material and other components of the waste, leading to the production of landfill gas and leachate thus waste conversion processes in LFB were studied and detailed. Effect of environmental factors mainly moisture content, pH, temperature, inhibitory influences and toxic components that affect the degradation processes in landfill bioreactors were also studied. Waste pre-treatment, co-digestion with other wastes, aeration, leachate management, LFG generation and extraction and reactor configurations are discussed as means to improve and steer the operation of LFBs. From this detailed study information was gathered that could be used in the set-up, development and evaluation of four innovative LFB system options. Especially information was collected regarding waste conversion rate, production of leachate, landfill gas generation rate, waste settlement and consolidation and stabilization of the waste.

Findings from the desk study revealed that waste degradation in conventional landfills can be enhanced by operating them as an anaerobic bioreactor. The underlying principle of the landfill bioreactor is that by optimizing operational control and environmental conditions within the waste by way of recirculation of leachate, more rapid and complete biodegradation of municipal solid waste may be achieved.

Table 8-1 is a summary of the various aspects that are relevant in the introduction of LFB technology to East Africa and their respective descriptions. Included in the table are also the benefits that can be realized in comparison with existing landfill operations currently practiced in East Africa. The aspects mentioned in the table include to operate the LFB in anaerobic mode with the crucial benefit of biogas production as a result of biodegradation of organic matter which is the major component in the MSW generated in East Africa. Another aspect is the introduction of pretreatment of MSW in a BIOCEL-system. The BIOCEL-system is a proven technology with capability of biogas production from the rapidly biodegradables in MSW in a short time (about 20 days) and also with capability for waste volume reduction as a result of the rapid biodegradation. The benefits are such as the use of LFB enhances stabilization of waste in a shorter time, efficient utilization of landfill capacity, more and rapid LFG (biogas) production, greenhouse gas emissions avoidance, control of odor, reduced leachate treatment costs and reduced post closure care of the landfill

Table 8-1: Relevant aspects regarding the application/implementation of LFB technology to East Africa

Aspects	**Description**
Feasible MSW Treatment Technology	Anaerobic treatment
Waste pretreatment technology • BIOCEL-system	Substantial amount of biogas production at short time
	Less volume of waste for final treatment
	No loss of biogas
Waste treatment technology • Standard landfill bioreactor (LFB)	Enhanced stabilization in a shorter time (10 years)
	Efficient utilization of landfill site
	Reduction of post closure care
LFB • Layout • Operation	>10 m waste height in a cell
	1 cell filled per week
	5 years of cell fully active (leachate recirculation, gas collection)
	After 5 years cell is partially active (no leachate recirculation, gas collection)
	After 10 years cell is completely closed (no leachate collection, no gas collection)
Landfill gas (biogas)	Active gas production and collection
	GHG Emission avoidance
Leachate management	Collection of all produced leachate
	Vertical wells leachate recirculation system
	In situ leachate treatment via recirculation
	Reduction of leachate treatment costs
	Ex-situ leachate treatment

8.2.3 Leachate management

Leachate production is a crucial aspect of the LFB so an in-depth desk study into leachate production was conducted. Generation of leachate remains an inevitable consequence of the existing landfilling practice and the future LFBs. The generated leachate needs to be treated to meet the standards for its discharge into municipal sewers or direct disposal into surface water. The LFB requires specific management activities of leachate and operational modifications to ensure enhanced biodegradation processes and in-situ leachate treatment are fostered. A brief study on leachate production, characteristics and recirculation in relation to LFBs was conducted. The proposed leachate treatment techniques are derived after a study on the world wide available technologies classified into the following major groups:

- leachate transfer to a municipal wastewater treatment plant
- aerobic and anaerobic processes
- chemical and physical methods
- membrane filtration: microfiltration, ultrafiltration, Nano filtration and reverse osmosis

In conclusion four leachate treatment options have been put forward and important leachate characteristics target pollutants of removal being some amount of BOD still in the leachate, persistent COD and ammonia-nitrogen. The LFB leachate treatment options proposed suitable for Tanzania, East Africa for the next 10-15 years are:

- Activated sludge process coupled with constructed wetland
- Pond system (facultative ponds) coupled with constructed wetland
- Evaporation
- Combined biological and physico-chemical treatment (Sequencing Batch Reactor-Coagulation/flocculation-Fenton oxidation process-aerobic biological treatment).

The first three proposed options are made bearing in mind the Tanzania, East African conditions (developing - tropical country) which are limited financial resources, inadequate present skills and capacity to manage sophisticated technologies, high temperature, medium rainfall, medium flow and high space availability, and target pollutants of removal. The fourth is a more process oriented and more sophisticated treatment option combining biological and physico-chemical treatment, based on batch reactor-Coagulation/flocculation-Fenton oxidation process-aerobic biological treatment and was also elaborated. However, this option is not appropriate for the current East Africa context and conditions.

8.2.4 Modified LFG production models

A landfill is a very complex heterogeneous environment and landfill processes are almost impossible to analyze in a deterministic way thus present considerable modeling challenges. LFG generation is a crucial impact that LFBs bring into the fray for decision makers to make a choice of the technology. Several methods have been described for modeling LFG production. Researchers began model development for prediction of gas recovery for both sanitary landfills and landfill bioreactors but the type of landfill considered was not always clear whereby different gas production models give very different results with individual landfills, even when the same data is entered. An overview of selected LFG models was conducted. The models include LandGem model (USEPA), First order model (TNO), Multiphase model (Afvalzorg), GasSim model (Golder Associates 2010 for the Environment Agency), French E-PRTR model and EPER model France (ADEME). These models in common assist in the determination of either landfill gas production rate or methane generation potential or methane production rate but not all models consistently use the same input parameters. There are some models that are dealing with organic carbon while others dealing with amount of organic matter or the total amount of waste. In some instances there are models that incorporate either dissimilation factors or sometimes conversion factor of carbon content in the waste. As a result of this overview more harmonization and some adaptation has been made to the models and the product being a modified innovative single phase model and a modified innovative multiphase model.

The modified single phase model is given by equation (8-1).

$$Q = M_o * X_o * f * \left(1 - e^{-k*t}\right) \ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots (8-1)$$

Where:

Q total amount of LFG (m^3)
M_o initial amount of waste deposited (ton)
X_o initial biodegradable organic matter fraction in the waste (kg OM/ton waste)
k adapted first order reaction rate constant ($year^{-1}$)
t time elapsed since deposition (year)

f conversion factor (m^3 LFG/kg OM converted)

The single phase model is further modified to a multiphase model through including the following differentiations:

- Three different organic fractions with different biodegradation rates are assumed to be present in the waste (slowly, moderately and rapidly biodegradable)
- Three first order reaction rate constants for the three different organic fractions are assumed
- Three different conversion factors for potential landfill gas production per ton of waste are assumed.

These innovative models can be easily understood and be used in the calculations of LFG production as used to quantify the amount of LFG produced for the operational years of the innovative LFB system options presented in this thesis.

8.2.5 Experimental research of pilot - scale anaerobic LFB in Tanzania

There are no operating landfills in East Africa that have been designed and operated as bioreactors, which means there is limited data on the knowledge and performance of LFBs. An exploratory research on a pilot scale landfill bioreactor filled with Tanzanian waste and leachate recirculation was executed. This pilot experiment was conducted under the prevailing environmental conditions of Tanzania, East Africa, to study the effect of recirculation on waste degradation and acidification, landfill gas production, and in situ leachate treatment and to provide insights for the successful operation of LFBs in developing countries. It was shown that acidification of the leachate in the LFB without production of LFG during a certain period is possible and that the LFB can be used for the first two steps of anaerobic digestion (i.e. hydrolysis and acidification) and then the remaining step of methanogenesis can be carried out in a separate reactor to produce biogas at a shorter period. It was also shown that the biogas production in the reactor with recirculation of leachate strongly increases the total biogas production compared to the reactor with no recirculation of leachate.

8.3 Proposed Landfill Bioreactor innovative options

Four innovative options of landfill bioreactor (LFB) systems feasible and adaptable in East Africa based on the use of the elaborated knowledge and pilot scale experimental results have been elaborated. The elaboration presents a schematic design and presentation of operation conditions of the LFB options and description of various unit operations and processes and an evaluation of the new ideas for sustainable MSW disposal or management for East Africa. The innovative concepts comprise of materials recovery and transfer stations (MR-TF). All the collected MSW before reaching the landfill site must be taken to the MR-TF. The sizing of the MR-TF is based on the population to be served or the tonnage of waste to be handles per year. Based on population, the envisaged MR-TFs are aimed at serving in areas with at least 100,000 people within a locality and spatially distributed covering the whole city's MSW collection areas. Alternatively the MR-TF should receive at least 100,000 tons wet waste per year thus for a city like Dar es Salaam, Tanzania, East Africa will have 10 MR-TFs based on the current MSW generation rate of an estimated 3000 tons/day.

The innovative system options also comprise of large scale centralized LFB and other supporting reactors for waste degradation such as the BIOCEL-system, leachate collection, storage tank and recirculation system, UASB reactor for leachate pre-treatment and ex-situ biogas production and leachate final treatment options and gas collection system.

The four developed innovative system options for treatment of waste that have been elaborated for treatment of waste in a LFB are considered as technically feasible. These system options are:

- System option 1: Standard LFB, focused on production of LFG from the LFB only.
- System option 2: Standard LFB with leachate acidification and LFG production from this acidified leachate in a UASB reactor, followed by LFG production from the LFB
- System option 3: Two stage treatment – first production of biogas from a centralized BIOCEL-system followed by production of LFG from the LFB
- System option 4: Two stage treatment – first production of biogas from a decentralized BIOCEL-system at MR-TF followed by LFG production from the Standard LFB

We adopted some standard conditions and assumptions for calculations to enable comparison and evaluation of the system options proposed in this thesis as summarized in Table 8-2. The required information for calculations are derived from the empirical research, literature compilation and exploratory experiments conducted throughout the research period.

Table 8-2: Standard conditions and assumptions made for calculations

Description	**Value**
Composition on dry basis	65% Biodegradable organic 35 % Inert (non-biodegradable) • 29% Inert organic • 6% Inert inorganic
Composition on wet basis (1 ton)	640 kg Water 360 kg dry matter
Distribution of the dry matter	234 kg (i.e. 65% Biodegradable organic) 104.4 kg (i.e. 29% Inert organic) 21.6 kg (i.e. 6% Inert inorganic)
At MR-TF 6% of the waste is removed as (dry) inert waste (60 kg)	53 kg inert organic (textiles and plastics) 7 kg inert inorganic (metals and glass)
Biodegradable organic fraction (1 ton) Slowly Moderately Rapidly	 25% (62 kg/ton) 42% (105 kg/ton) 33% (82 kg/ton)
Assumptions made: Density of the waste Gas composition LFG collection efficiency Oxidation efficiency by top cover BIOCEL-system biogas potential	 0.5 ton/m^3 50% CH_4; 50% CO_2 at STP 80% 50% 70 kg biogas/300 kg OM

Using the basic data on LFB from chapters 2-6 (summarized in Table 8-1) and adopted standard conditions and input data (Table 8-2) we calculated the LFG potential from 1,000,000 collected waste and after removal of 6% recyclables at the MR-TF leaving 940,000 tons/year wet waste for disposal which amounts to 168,480,000 m^3/year. Using the modified LFG production models (chapter 5) the annual LFG production based on 10 years of active operation by the various system options (at the set standard assumptions) is 125 million m^3

(about 74-75% of the potential amount) for all System options. The greenhouse gas emissions (net methane emissions) from the System options 1, 2, 3 and 4 are 10.2%, 9.2%, 8.9% and 8.9% respectively whilst from existing controlled dump the net emission (i.e. 25%) is more than double the emission by the system options.

Between the proposed system options, the system options with BIOCEL-system have a lesser net methane emissions than the other two options as shown in Table 8-3.

Table 8-3: Summary of system options showing the volume of LFG collected, GWF, LFB size and BIOCEL size (10 years operational time)

System options	Option 1	Option 2	Option 3	Option 4
LFG collected (m^3)	99,868,867	103,370,470	108,582,132	108,582,132
CH_4 emitted (m^3)	17,152,783	15,516,833	14,974,467	14,974,467
GWF (Tons CO_2- eq. per year)	259,350	234,615	223,692	223,692
LFB site (ha/year)	18.8	18.8	15.4	15.4
BIOCEL size (ha)	n/a	n/a	3.364	0.3364 * 10

Greenhouse gas emissions from the system options are more are less identical with the range of 224,000 – 260,000 tons CO_2. eq. per year

Land space requirement of the site for the innovative system options is 18.8 ha/year for System option 1 and 2 and for system option 3 is 15.4 ha/year for the LFB plus a one-time land space of 3.364 ha for the centralized BIOCEL-system. For option 4 the size of the LFB is same as that of System option 3 that is 15.4 ha and for the decentralized BIOCEL-systems is 0.3364 ha under the assumption that there will be 10 of such systems located at 10 MR-TFs

Active operation of 10 years is more or less an adequate length of time for the Standard LFB because by that time at least 74% of the potential LFG is already produced which is a significant amount.

Amount of leachate envisaged to be produced and requiring treatment is the same for all system options which is 589,354 tons/year. For System option 4 part of the leachate is produced during the 20 days of biodegradation in the BIOCEL-system at the decentralized MR-TF then it is transported to the landfill site.

Separate partial drainage of water from the waste

The waste in East Africa is characterized by a moisture content of about 60% which is very high and which is significantly above field capacity. An alternative to the water and leachate management as described in the four innovative system options is to drain part of the water content from the waste before deposition of the waste in a LFB or BIOCEL-system. It is expected that it is easily possible to drain the water in the waste that is above 50%. Then the waste that is fed to the LFB or BIOCEL-system has a water content of 50%. With this assumption we can calculate the amounts of water that have to be removed at the MR-TF and the LFB site. The results are presented in Table 8-4. The resulting consequences due to separate partial drainage of water from the waste are:

- Smaller landfill sites for eventual waste deposition
- Smaller reactor size – BIOCEL reactors
- Less leachate to be handled and treated
- Less waste to be transported

Table 8-4: Amount of water removed and the resulting size requirement

System option	Amount of water removed		
	MRTF (tons leachate/year)	BIOCEL (tons leachate/year; Size)	LFB (tons leachate/yr; Size)
1	340,000	n/a	249,440; 12 ha
2	340,000	n/a	249,440; 12 ha
3	340,000	54,000; 2.208 ha	195,000; 9.84 ha
4	34,000*	5,400; 0.2208 ha*	195,000; 9.84 ha

*- The amount is for each decentralized BIOCEL-system at 10 MR-TF

Sensitivity analysis

A number of assumptions have been made for the calculation and comparison of the four system options. However, these assumptions can raise questions regarding the effectiveness, capacity or performance of the various system options and regarding the conclusions about the comparison of the various systems. Therefore a sensitivity analysis was performed on most of the assumed values and a comparative analysis was evaluated. Alteration of the following parameters were included in the sensitivity analysis:

- Length of active operation time by including 5 and 15 years following the LFG production, LFG collection and its respective GWF
- Collection efficiency in LFB at a level of 75% and 85%
- Variation of the acidification regime by including 0 years, 0.5, 1 and 2 years on the 10 years active operation time and 80% collection efficiency
- Change of the assumed 70 kg gas produced from 300 kg OM present in the biowaste for calculation of the amount of biogas produced in the BIOCEL-system. Now including the assumption of biogas production of 70 kg biogas produced from 200 and 400 kg OM
- 5 years operation time, 75% collection efficiency, 25% oxidation by top cover and 70 kg biogas produced in the BIOCEL-system from 200 kg OM

From the sensitivity analysis the following conclusions can be drawn:

- The amount of LFG collection and greenhouse gas emissions is not very sensitive for small changes in collection efficiency of the LFG, active operation time at the site and acidification regime of System option 2. However, acidification of the leachate in the new cells for System option 2 may require more manageability aspects but is particularly helpful when you have to deal with a low recovery efficiency of the LFG from the LFB
- Application of a BIOCEL anaerobic reactor has a strong influence on the size of the site, and also on the emission of greenhouse gases if the gas collecting efficiency is low and the conversion factor of the methane in the biogas that is not collected is low.

A qualitative cost analysis was also made and it was revealed that all four system option cost more than the controlled dump by the addition of the leachate collection and recirculation system and the gas collection system. System option 3 and 4 with the BIOCEL-system is even much more costlier than all the other options but the investment costs are offset by the utilizing LFG that can be a source of monetary benefits via the production of electricity and by claiming the CDM credits from the net avoided emissions which is a strength of the System option with the BIOCEL-system.

8.4 Critical look back

To achieve the set objectives and answering the research questions a number of activities were carried out, methods employed, studies conducted, assumptions made, calculations performed and experiments were carried out. In this thesis four innovative system options for treating MSW in East Africa by means of landfill bioreactors have been developed, elaborated and evaluated. For the evaluation we have used mathematical models. Based on input parameters these mathematical models can calculate for each option the amount of biogas that is produced, collected, the emission of methane to the atmosphere, the size of the landfill bioreactor and the size of pre-treatment systems. Some of these input parameters are input which is dealing with strictly defined conditions such as amount of waste that goes to the pre-treatment system, amount of waste that goes to the disposal site, active retention time of the waste in the disposal site (active operation time in the disposal site). These input parameters can be changed independently. However several parameters are dealing with the composition of the waste and the conversion processes that takes place in the site or in the waste pre-treatment system. Data for these parameters have been deduced partly from own research, such as the composition of the waste and the production of biogas from the waste, and partly from literature data dealing with sanitary landfills and landfill bioreactors from Western Europe or the U.S.A. However, the latter group of data deals with specific waste from Western countries that is different from East Africa with respect to composition and relevant properties. All these aspects had some uncertainties, speculative inferences and in some instances lack of data to make conclusive remarks. The challenges that we faced during this study were centred around the following critical factors which are crucial for the evaluation of the proposed system options for East Africa. These factors include:

- Amount of biodegradable organics present in the waste of East Africa
- Amount of water present in the waste of East Africa
- Biodegradation rate constants of the organic waste of East Africa. For the various types of organic waste. With respect to the production of biogas or to the acidification process
- Conversion efficiencies of OM to biogas
- BIOCEL-system (and other relevant systems) performance in tropical countries
- Collection efficiency of LFG generated from the LFB
- Oxidation efficiency by the top cover of the LFB
- Field capacity of the waste in the bioreactor the LFB

Furthermore all these data show a scattering and sometimes also lacking consistency. From the calculations and the sensitivity analysis we have seen that the specific composition and properties of the waste a have strong influence on the technical and economic performance of the four innovative system options. It means that for a more accurate calculation of the performance of the four innovative system options more accurate input data is necessary. Input data that represents more accurately the typical conditions in East Africa. More accurate parameters which can be obtained through experimental research or even full scale experiments are recommended to make better and more reliable calculations and draw more elaborative conclusions particularly on the LFG potential, emissions of greenhouse gases and the length of time during which the acidification regime of the leachate can be maintained.

8.5 Final conclusions and recommendations

8.5.1 Final conclusions

1. Municipal Solid Waste (MSW) collected in East African cities is characterized dominantly by a high content of organic material and a moisture content of above 60%. Most common disposal option currently practiced in most or all East African cities is controlled dumping geared towards landfilling. Landfilling is an essential part of an integrated waste management strategy, without which effective waste management will not be possible. It is expected that more sophisticated and modern forms of landfill such as a Landfill Bioreactor (LFB) will become important treatment system for MSW in East Africa on the short and middle term.

2. Based on literature information regarding the construction and performance of Landfill Bioreactors in highly industrialized western countries, characteristics of MSW in East Africa, experimental research on pilot plant scale and desk studies regarding biological conversion, modeling of the biodegradation rates and biogas production of MSW four innovative options (System options) of Landfill Bioreactors were identified, elaborated and evaluated. All system options are based on a combination of decentralized collection and partial treatment of the MSW at materials recovery and transfer stations (MR-TF) and transport from these MR-TF to a landfill bioreactor disposal site.

3. The four options are:

   a. System option 1: Standard LFB, focused on production of LFG from the LFB only.
   b. System option 2: Standard LFB with leachate acidification and LFG production from this acidified leachate in a UASB reactor, followed by LFG production from the LFB
   c. System option 3: Two stage treatment – first production of biogas from a centralized BIOCEL-system followed by production of LFG from the LFB
   d. System option 4: Two stage treatment –first production of biogas from a decentralized BIOCEL-system at MR-TF followed by production of biogas from the Standard LFB

4. These four options were evaluated by means of a semi-mathematical calculation model for their investment and operation costs, land space requirement, leachate treatment costs and savings, LFG generation and LFG collection and utilization costs and benefits, airspace recovery and greenhouse gas accounting and global warming avoidance. The main results as presented in Table 8-5 with respect to this evaluation are compared with a controlled dumpsite for MSW as currently applied in East Africa, all four modifications of the LFB show great advantages with respect to landfill size, amount of biogas collected and reduction of the emission of greenhouse gases.

Table 8-5: Summary of costs and benefits for the System options and Control dump as existing situation

System option	Investment and O&M Costs: Land value, landfill preparation and operation	Waste pre-treatment	Leachate treatment savings	Potential monetary benefits: Utilizable LFG generation of the potential amount	GHG accounting/GWF
Controlled dump	Basic costs **18.8 ha/year**	None	None Standard amount requiring full treatment	None	25% net CH_4 emissions **Emitted 635,040** Ton CO_2-eq./yr.
Standard LFB Option 1	Basic costs **18.8 ha/year**	Additional (i.e. costs of the MR-TF)	Less leachate (Polishing)	59.3%	10.2% net CH_4 emissions **Emitted 259,350** Ton CO_2-eq./yr.
LFB+UASB Option 2	Basic costs **18.8 ha/year** + Space for UASB	Additional (MR-TF)	Less leachate	63.1%	9.2% net CH_4 emissions **Emitted 234,615** Ton CO_2-eq./yr.
BIOCEL+LFB Option 3	Basic costs **15.4 ha/year** + Space for BIOCEL	Additional + MR-TF + BIOCEL system costs	Less leachate Lower cost of leachate treatment	64.4%	8.9% net CH_4 emissions **Emitted 223,692** Ton CO_2-eq./yr.
Decentralized BIOCEL+LFB Option 4	Basic costs **15.4 ha/year** + Space for decentralized BIOCEL reactors	Additional + MR-TF + decentralized BIOCEL-system costs	Less leachate (Polishing)	64.4%	8.9% net CH_4 emissions **Emitted 223,692** Ton CO_2-eq./yr.

5. It was pointed out that the outcome of the results of the calculation are sensitive for the used input variables. Because most of these input variables have been derived from evaluation of LFBs in highly industrialized Western countries, there is a strong need to verify to what extent these variables are sufficiently characteristic for the typical situation of MSW in East Africa

6. The used semi-mathematical model is very flexible with respect to input variables and extension of the whole treatment chain with additional treatment steps.

7. Important economic and technical benefits in the treatment process of MSW can be achieved if the MSW that is collected, has a lower water content. In that respect especially policy measures might be introduced which stimulate people to minimize the water content in the MSW as much as possible

8.5.2 Recommendations

- More research need to be conducted to gather relevant information about leachate characteristics emanating from East African waste, leachate generation rate and leachate treatment.

- There are a number of assumptions made for the calculation and evaluation of the system options. These assumptions need further quantification and research to delete any uncertainties in the results before implementation of the technology at full scale
- Trial BIOCEL-system for performance evaluation in tropical conditions needs to be conducted
- Optimize technically and financially the integrated treatment of the MSW at the MR-TF stations and at the LFB site

CHAPTER 9

Summary

9.1 Introduction

This thesis takes as point of departure of the need of cost-effective, land-saving and energy producing waste treatment technologies for East African cities. The main objective of the thesis is to develop and describe landfill bioreactor based municipal solid waste (MSW) treatment systems suitable for East African cities. For this purpose, four innovative landfill bioreactor system options which are technically feasible and resource-recovery oriented that match the conditions of East-African cities are developed. The innovative system options proposed in this thesis and the elaborations together with the evaluations are useful and helpful for decision makers in making the choice of MSW disposal suitable for the cities in East Africa.

This thesis is comprised of nine chapters. chapter 1 introduces the study context, research objectives and research questions. It provides information on the problems of MSW management in Tanzania and East Africa and explains the rationale of the thesis. Chapter 2 presents findings from an empirical research conducted in Mwanza City as an exemplary case study focusing on the current MSW management practice in East Africa, waste characteristics and generation. Chapter 3 is a review of the LFB, general design of the LFB, the processes involved and steering parameters that influence operation of the LFB. Chapter 4 describes the available technological interventions of leachate for optimal management and treatment of the residues in the leachate after pre-treatment or recirculation in LFB, UASB reactor and the BIOCEL process. Chapter 5 presents an overview of existing models for gas production and presents simplified innovative models for ease of calculation of waste degradation and gas production in LFBs. Chapter 6 describes the performance of a pilot scale LFB experiment conducted in East Africa to evaluate the effect of recirculation on waste degradation and acidification, landfill gas production, and in situ leachate treatment The outcomes of chapters 2-6 are applied in chapter 7. Chapter 7 reveals innovative LFB configurations aimed at optimization of energy recovery and suitable for the East African context. Chapter 8 presents findings, critical look back at the evaluation and elaboration of the system options and finally conclusions and recommendations. Chapter 9 summarizes the answers to the research questions raised in chapter 1.

9.2 Synopsis

Chapter 2 investigates the current conditions and practices of waste collection and disposal in East African cities. The main objective is to find a basis for improved waste management in East Africa by diagnosing the current MSW management practice in one of the major cities of the country in the Lake Victoria region (Mwanza City in Tanzania). Through diagnosis of MSW management which included mainly waste characterization and making an inventory for the current practices in waste collection and disposal in the study area. From the study component on characterization, it was found that the collected waste has a high organic content, that is 84% of all wastes is organic in nature while 14% is amenable to recycling and reuse such as papers and boxes, plastics, metals and glass whereas other materials such as e-waste, batteries, ceramics, etc. are 2.7%. Without proper attention to the biodegradable fraction of waste such as appropriate landfilling technologies of the waste, environmental pollution, health and degradation implications will be imminent. The information on waste characteristics and composition was used for appropriate choice of systems to manage the waste and needed for the elaboration of suitable system options for Tanzania in chapter 7 of this thesis. Findings from the diagnosis revealed collection and disposal being the main MSW functional problems in East Africa. In order to curb the problems related to collection, City authorities have opted for privatization of the services and significant improvement was realized. The amount of waste collected for disposal before privatization was less than 40% and as a result of privatization in 2002, the collection efficiency rose to 61% just after 2 years of operation (i.e. in 2004) and the collection efficiency was observed to be on a gradual rise. The method of disposing of waste in Mwanza City is landfilling, in fact uncontrolled dumping. Like Mwanza, Nairobi has one designated waste disposal site, an open dump located in Dandora area, Kampala city has a sanitary landfills at Kitezi and Dar es Salaam city at Pugu Kinyamwezi the disposal site is designed as a sanitary landfill but has been implemented as a controlled dumpsite.

In these cities, the waste collected is predominantly organic thus presenting opportunities for: enhanced stabilization of the organic fraction of the waste which is clearly the largest portion of the generated waste; potential for landfill gas recovery; reduced leachate treatment potential. This calls for an urgent need for improving the disposal practices that these East African countries are carrying out.

Chapter 3 presents a literature review of the fundamental processes in a landfill bioreactor, the design of a landfill bioreactor and operation of landfill bioreactors. It aims to generate the insights to make well founded choices about the introduction of the landfill bioreactor technology in East Africa. It is a review of the LFB, general design of the LFB, the processes involved, steering parameters that influence operation of the reactor and closure and post closure issues to be addressed. The development of a more sustainable landfill is important to the safe and effective management and control of municipal solid waste in the future. The concept of landfill bioreactors technology is relatively very new to developing countries like Tanzania in East Africa. Table 9-1 is a summary of the various aspects that surround the introduction of LFB technology to East Africa and their respective descriptions which also include benefits that can be accrued by implementation.

Table 9-1: Basic information and benefits for choice of applicability of LFB in East Africa

Aspects	**Description**
Waste pretreatment • BIOCEL-system • Drainage of a substantial part of the water present in the MSW (at MR-TF or at LFB site)	Biogas production at short time
	Less volume of waste for final treatment
	Less waste for treatment in BIOCEL reactors
	Less waste for treatment in the LFB
Waste treatment advantages with respect to Standard landfill bioreactor	Less leachate finally to be treated at the site
	Enhanced stabilization in a shorter time (10 years)
	Efficient utilization of landfill site
	Reduction of post closure care
	More biogas
LFB Cell • Layout • Operation	less Greenhouse gases
	>10 m cell depth
	Cell filling on weekly basis
	5 years of cell activity (leachate recirculation, gas collection

	After 5 years cell is partially active (no leachate recirculation, gas collection)
Landfill gas (biogas)	After 10 years cell is completely closed (no leachate collection, no gas collection)
	Gas collection
Leachate management	GHG Emission avoidance
	Collection of produced leachate
	Vertical wells leachate recirculation system
	In situ leachate treatment

The aspects include to operate the LFB in anaerobic mode with the crucial benefit of biogas production as a result of biodegradation of organic matter which is the major component in the MSW generated in East Africa. Another aspect is the introduction of pretreatment of MSW in a BIOCEL-system. The BIOCEL-system is a proven technology with capability of biogas production from the rapidly biodegradables in MSW in a short time (about 20 days) and also waste volume reduction as a result of the rapid biodegradation. Other aspects include modalities of cell operation (filling time, length of active period, depth), leachate and gas collection systems, leachate recirculation and in-situ treatment. Included in the table is also the benefits that can be realized in comparison with existing landfill operations currently practiced in East Africa. The benefits are such as the use of LFB enhances stabilization of waste in a shorter time, efficient utilization of landfill capacity, more and rapid LFG (biogas) production, green gas emissions avoidance, control of odor, reduced leachate treatment costs and reduced post closure care of the landfill.

Chapter 4 looks at the available treatment technologies of leachate for optimal management and treatment of the leachate after pre-treatment or recirculation in LFB, UASB reactor and BIOCEL process. A literature review has been made on the management of leachate emanating from appropriate technology applied for treatment of MSW in East Africa that is advocated for in this thesis, namely the LFB with a cross reference with sanitary landfills. Much reference is made to sanitary landfill because most data about MSW and leachate management in the literature refer to sanitary landfill, much less has been published about the specific leachate quality, quantity and treatment related to LFBs. Generation of leachate remains an inevitable consequence of the existing landfilling practice and the future LFBs. The generated leachate needs to be treated to meet the standards for its discharge into municipal sewers or direct disposal into surface water. Bearing in mind the Tanzania, East African conditions (developing - tropical country) which are limited financial resources, inadequate present skills and capacity to manage sophisticated technologies, high temperature, medium rainfall ,medium flow and high space availability, and target pollutants of removal being the primarily persistent COD and relatively high ammonia-nitrogen concentrations and absence of VFA and some BOD then aerobic treatment and lagooning techniques are proposed as suitable for East Africa for the next 10-15 years. Therefore we put forward 4 options:

- Activated sludge process coupled with a constructed wetland;
- Pond system (facultative ponds) coupled with a constructed wetland;
- Evaporation;
- Combined biological and physico-chemical treatment.

The first three proposed options are made bearing in mind the current Tanzania, East African conditions (developing - tropical country) which are limited financial resources, inadequate present skills and capacity to manage sophisticated technologies, high temperature, medium rainfall , medium flow and high space availability, and target pollutants of removal. The fourth is a more process oriented and more sophisticated treatment option, based on batch reactor-Coagulation/flocculation-Fenton oxidation process-aerobic biological treatment was also elaborated. However, this option is not appropriate for the current East Africa context and conditions but rather for future.

Chapter 5 presents an overview of models for calculation of the waste degradation and gas production in LFBs. The outcomes of this chapter are applied in chapter 7. The focus of this chapter is on modeling of landfill gas (LFG) generated as a result of waste landfilled whereby the organic fraction in the waste decomposes. In this chapter a literature review encompassing models for quantification of methane generation from a landfill bioreactor and a critical evaluation of the models is discussed. There exist several models from various literature sources. All the models are based on a first order biogas production rate in the total amount of waste, the organic matter content of the waste or the organic carbon content of the waste. These models in common assist in the determination of either landfill gas production rate or methane generation potential or methane production rate but not all models consistently use the same input. In this thesis we have presented an improved modified model of biogas generation. We have applied this general modified model for the calculations of biogas production from one cell and a series of many cells with waste landfilled over a specified period of time. This improved modified model can be easily understood and was used in chapter 7 of this thesis.

Chapter 6 reports on findings from a comparative study of a pilot scale landfill bioreactor. This pilot scale experiment was conducted in Dar es Salaam city, Tanzania, East Africa, to study the effect of recirculation on waste degradation and acidification, landfill gas production, and in situ leachate treatment. In order to achieve this objective the following activities were undertaken: (1) study of the variations of the effluent leachate characteristics as an indicator of waste stabilization, (2) evaluation of the effects of leachate recirculation on leachate COD removal, (3) evaluation of the landfill gas generation rate and composition, (4) monitoring of the settlement of waste due to the organic matter degradation. The pilot setup consisted of two reactors without (R1) and with (R2) leachate recirculation. R1 is operated as a control reactor simulating a sanitary landfill and R2 is considered as a simulated landfill bioreactor. Each of the reactors were simultaneously filled with about 2.3 tons of wet waste of moisture content of about 64%. Throughout the study of 52 weeks R1 was run as a flow-through system whereas R2 was broken into two phases. During phase one of reactor R2 the leachate was recirculated directly to the top of the reactor and phase two involved recirculation of leachate after treatment via an Up-flow Anaerobic Sludge Blanket (UASB) reactor as an in-situ pre-treatment measure of the leachate. The UASB reactor used was a 15.7 litre PVC reactor of 2 m height, 0.1 m diameter, HRT of 1.15 days, filled with 6.75 L anaerobic sludge obtained from an existing UASB reactor whose sludge age is more than 5 years with a specific methanogenic activity of about 0.17 g COD/g VSS/day. The main results of this study indicate the validity and feasibility of operation of the LFB with waste characteristics of East Africa to accelerate the stabilization of organic-rich wastes, enhance LFG production and achieve a degree of leachate treatment. From the study specific conclusions drawn are: confirmation of the feasibility of the operation of a landfill as a controlled anaerobic bioreactor with leachate recirculation; leachate recirculation enhanced waste stabilization as reflected in higher gas production in R2 (simulated LFB) than in R1 (control) and more waste settlement; Controlled acidification of the leachate is possible; In practice, the two stage approach of extended acidification means that no biogas is generated within the landfill so that there is no loss of methane from the landfill. Accordingly the two-stage process may result in a lower overall loss of biogas to the atmosphere and; management of nutrients (N and P) requires attention because neither degradation nor removal of these parameters was observed in both R1 (control) reactor and R2 (simulated LFB).

Chapter 7 presents four innovative concepts called system options of landfill bioreactor (LFB) systems adaptable in East Africa making use of advanced existing knowledge and pilot- scale experimental results. The chapter also presents schematic design and operation of landfilling system options. The developed concepts comprise of materials recovery and transfer stations (MR-TF). At the transfer station, recovery of materials is established where non-biodegradable materials are sorted and removed from the waste input stream of the LFB or the BIOCEL-system. The concepts also comprise of, large-scale LFB and other supporting reactors for waste degradation, such as the BIOCEL, leachate storage tank and recirculation system, leachate primary and final treatment and gas collection systems. The BIOCEL process, leachate recirculation, various leachate treatment options and gas collection systems have been thoroughly reviewed in chapters 3 and 4. In chapter 5 the models to calculate biogas production have been elaborated. The development of these innovative system options has been achieved through the following: a) empirical research conducted in East Africa - Tanzania to diagnose the existing MSW management practices and amount and characteristics of collected MSW (chapter 2); b) extensive literature review on LFBs and leachate treatment (Chapters 3 and 4); c) gas production modeling (chapter 5) and d) a locally conducted pilot-scale study (chapter 6). The proposed LFB system options hinge on the waste matrix to be landfilled, the leachate generation, extraction and treatment and the location of LFG generation as the variables in the selection of the system options. The proposed systems are:

- System option 1: Standard landfill bioreactor;
- System option 2: Standard LFB with part of LFG production in a UASB reactor at the LFB site;
- System option 3: Two-stage treatment – Centralized BIOCEL followed by a LFB;
- System option 4: Two-stage treatment – Standard LFB fed by decentralized BIOCEL reactors at transfer stations

By means of a semi- empirical model We adopted some standard conditions and assumptions for calculations to enable comparison and evaluation of the system options and calculated the amount of LFG that is produced, the emission of methane, the size of the landfill site, and the amount of leachate that is produced and that has to be treated. The waste in East Africa is characterized by a moisture content of about 60% which is very high and which is significantly above field capacity. We therefore assessed and evaluated an alternative to the water and leachate management as described in the four innovative System options that is to drain part of the water content from the waste before deposition of the waste in a LFB or BIOCEL-system and presented the resulting consequences. Furthermore, we carried out a sensitivity analysis on most of the assumed values and a comparative analysis was evaluated. From the sensitivity analysis amount of LFG collected and greenhouse gas emitted is not very sensitive for small changes in collection efficiency of the LFG; acidification of the leachate in the new cells may require more manageability aspects but is particularly helpful when you have to deal with low recovery efficiency from the LFB; and application of a BIOCEL anaerobic reactor has a strong influence on the size of the site, and also on the emission of greenhouse gases if the gas collecting efficiency is low and the conversion factor of the methane in the biogas that is not collected is low. More accurate parameters which can be obtained via experimental research are required to make better and more reliable calculations particularly on the biogas potential of the BIOCEL-system, oxidation efficiency of the top cover, recovery efficiency of the LFB, emissions of greenhouse gases and the length of time to which the acidification regime of the leachate can be achieved. All the four system options cost more than the conventional landfill due to the addition of the leachate collection and recirculation system and the gas collection system. System options with the BIOCEL-system is even much more costlier than all the other options but the investment costs can be offset by the utilizable LFG that can be a source of monetary benefits via the production of electricity and claiming the CDM credits from the net avoided emissions which is a strength of these System options with the BIOCEL-system. And the calculation model that has been developed can easily be applied with input parameters which are different from the standard input parameters. Together with the four system options the calculation model provides a useful tool for decision making regarding municipal solid waste management in East African countries.

Chapter is 8 presents a discussion and the final conclusions of the previous seven chapters of this thesis. The main objective of this thesis is to have developed and described landfill bioreactor based municipal solid waste treatment systems suitable for East African cities. In this chapter a critical look back at the evaluation and elaboration of the system options is made and finally the following main conclusions are put forward. (1) MSW collected in East African cities is characterized dominantly by a high content of organic material and a moisture content of above 60% and landfilling is an essential part of an integrated waste management strategy. It is expected that a more sophisticated and modern form of landfill such as a LFB will become important as a treatment system for MSW in East Africa on the short or middle term. (2) Four innovative modifications (System options) of landfill bioreactors were identified, elaborated and evaluated based on literature information regarding the construction and performance of landfill bioreactors in highly industrialized western countries and characteristics of MSW in East Africa, experimental research on pilot plant scale and desk studies regarding biological conversion, modeling of the biodegradation rates and biogas production of MSW. (3) These four options were evaluated by means of a semi-mathematical calculation model for their investment and operation costs, land space requirement, leachate treatment costs and savings, LFG generation and LFG collection and utilization costs and benefits, airspace recovery and greenhouse gas accounting and global warming avoidance. Finally, compared with a controlled dumpsite for MSW as currently applied in East Africa, all four modifications of the LFB show great advantages with respect to landfill size, amount of biogas collected and emission of greenhouse gases.

HOOFDSTUK 9

Samenvatting

9.1 Inleiding

Het uitgangspunt van dit proefschrift is de behoefte aan kosteneffectieve, ruimtebesparende en energie producerende afvalverwerkingstechnologieën voor Oost-Afrikaanse steden. Het hoofddoel van het proefschrift is het ontwerpen en beschrijven van een zgn. Bioreactorstort geschikt voor behandeling van het stedelijk afval van Oost-Afrikaanse steden. Ten behoeve van deze doelstelling is een viertal innovatieve Bioreactorstortsystemen ontworpen die technisch haalbaar worden geacht, gericht zijn op het terugwinnen van waardevolle componenten en die voldoen aan de specifieke condities van de genoemde steden. De innovatieve Bioreactorstortsystemen die in dit proefschrift worden voorgesteld, uitgewerkt en geëvalueerd zijn nuttig en ondersteunend voor instanties die moeten beslissen over de keuze van een stortsysteem voor stedelijk afval in Oost-Afrika.

Het proefschrift bestaat uit negen hoofdstukken. Hoofdstuk 1 geeft een introductie van de context van het proefschrift, de onderzoekdoelen en onderzoekvragen. Het geeft informatie over het managementprobleem van stedelijk afval in Tanzania en Oost-Afrika en verklaart de opzet en motivatie van het proefschrift. In hoofdstuk 2 worden de resultaten vermeld van een empirisch onderzoek dat is uitgevoerd in Mwanza City als een voorbeeldcasus van de huidige managementpraktijk inzake stedelijk afval in Oost-Afrika, de hoeveelheid afval die wordt geproduceerd en de karakteristieke samenstelling van het afval. In hoofdstuk 3 wordt op basis van een literatuurstudie een overzicht gegeven van het Bioreactorstort systeem, met name wat betreft het algemene ontwerp, de processen die erin plaats vinden en de stuurparameters die de werking van de Bioreactorstort bepalen. Hoofdstuk 4 beschrijft aan de hand van literatuuronderzoek de productie en samenstelling van percolatiewater, recirculatiesnelheden en technologieën voor een optimale zuivering van het percolatiewater, nadat dit voorbehandeld is middels recirculatie over de Bioreactorstort, middels behandeling in een UASB reactor of via recirculatie over een BIOCEL reactor. Hoofdstuk 5 geeft een overzicht van bestaande mathematische modellen voor de productie van stortgas en behandelt voorts eenvoudige mathematische modellen voor de berekening van de afbraak van het afval en de gasproductie in een Bioreactorstort. Hoofdstuk 6 beschrijft een onderzoek naar de werking van een Bioreactorstort op pilotschaal in Oost Afrika waarbij het effect van een recirculatie van percolatiewater op de verzuring, de productie aan stortgas en de zuivering van het percolatiewater zijn onderzocht. De resultaten van dit onderzoek worden toegepast in hoofdstuk 7. In hoofdstuk 7 worden enkele innovatieve modificaties van het Bioreactorstort systeem behandeld die gericht zijn op optimalisatie van de energieproductie en waarvan de toepassing binnen de Oost-Afrikaanse context mogelijk wordt geacht. Hoofdstuk 8 geeft de belangrijkste resultaten weer, kijkt nog eens kritisch terug op de uitwerking en evaluatie van de diverse opties van het Bioreactorstort systeem en vermeldt de eindconclusies en aanbevelingen. Hoofdstuk 9 geeft een samenvatting van de antwoorden op de onderzoekvragen die in hoofdstuk 1 zijn vermeld.

9.2 Overzicht

In hoofdstuk 2 wordt de huidige situatie en context met betrekking tot de praktijk van de afvalinzameling en afvalverwerking beschreven. Het hoofddoel is een basis te vinden voor een verbetering van het afvalmanagement systeem in Oost-Afrika op basis van onderzoek naar de huidige management praktijk inzake stedelijk afval in een van de grote steden in het gebied van het Victoriameer (Mwanza City in Tanzania). Dit onderzoek omvatte een afvalkarakterisering en een inventarisatie van de huidige praktijk van afvalinzameling en afvalverwerking. Uit het onderzoek naar de karakterisering van het afval kwam naar voren dat het ingezamelde afval een hoog gehalte aan organisch materiaal bevat. Het blijkt, dat 84 % organisch van aard is terwijl 14 % geschikt is voor recycling en hergebruik, zoals papier, dozen, plastic, metaal en glas. De rest van het afval, 2,7 % bestaat o.a. uit elektronisch afval, batterijen en keramisch afval. Zonder informatie over de biodegradeerbare fractie in het afval was het niet mogelijk om zicht te krijgen op mogelijk bedreigende factoren die het storten van afval met zich mee kan brengen zoals milieuvervuiling en gezondheidsproblemen. De informatie over de samenstelling van het afval is gebruikt om tot een juiste keuze van afvalmanagement systemen te komen en om managementopties die speciaal geschikt zijn voor Tanzania uit te werken in hoofdstuk 7. De onderzoekresultaten bevestigden dat inzameling en verwerking van stedelijk afval de belangrijkste problemen zijn met betrekking tot het functioneren van het afvalmanagementsysteem in Oost-Afrika. Teneinde het probleem van het inzamelen van afval aan te pakken hebben de stedelijke besturen rond het jaar 2000 gekozen voor de privatisering van de dienstverlening. Dit heeft geleid tot een aanzienlijke verbetering. De hoeveelheid afval die ingezameld werd vóór de privatisering was minder dan 40 %. Privatisering in 2002 leidde binnen 2 jaar tot een stijging van het inzamelpercentage tot 61 %. Dit inzamelpercentage vertoont een geleidelijke stijging. Het afval van Mwanza City wordt ongecontroleerd gestort. Evenals Mwanza beschikt Nairobi over een aangewezen stortlocatie, een open stort in het gebied van Dandora. Kampala City heeft een sanitaire stort in Kitezi. Dar es Salaam City beschikt over een stort in Pugu Kinyamwezi. Deze stort is ontworpen als een sanitaire stort maar wordt gebruikt als een gecontroleerde stort. Het afval dat in deze steden wordt ingezameld is overheersend organisch van aard hetgeen mogelijkheden biedt voor een versnelde stabilisatie van de organische fractie in het afval, mogelijkheid tot de winning van stortgas, en vereenvoudigde behandeling van het percolatiewater. Er is derhalve een dringende behoefte aan verbetering van de bestaande praktijken van het storten van afval die in deze Oost-Afrikaanse landen worden toegepast.

In hoofdstuk 3 wordt een literatuuroverzicht gegeven van de fundamentele processen die in een Bioreactorstort plaatsvinden, het ontwerp van een Bioreactorstort system en het bedrijven van een dergelijk systeem in de praktijk. Het doel is om inzichten te verkrijgen die het mogelijk maken om tot goed gefundeerde keuzes te komen bij de introductie van Bioreactorstort systemen in Oost-Afrika. Het overzicht besteedt in het bijzonder aandacht aan de parameters waarmee de werking van de Bioreactorstort kan worden beïnvloed alsmede aan de aspecten die van belang zijn bij de start van een dergelijke stort en bij het sluiten ervan. De ontwikkeling van een meer duurzame stort is van groot belang voor een veilig en effectief afvalmanagement in de toekomst. Het concept van de Bioreactorstort is relatief nieuw voor ontwikkelingslanden zoals Tanzania en andere Oost-Afrikaanse landen. In tabel 9-1 wordt een overzicht gegeven van de verschillende aspecten die relevant zijn bij de introductie van Bioreactorstort technologie in Oost-Afrika, alsmede een zeer globale beschrijving van deze systemen en een vermelding van de voordelen die kunnen worden verkregen bij implementatie van deze technologie.

Tabel 9-1: Basisinformatie over de toepassing van een Bioreactorstort in Oost-Afrika

Aspecten	**Karakteristieken**
Afvalvoorbehandeling • BIOCEL-systeem • Drainage van een substantieel deel van het water aanwezig in Stedelijk afval (op de overslaglocatie of op de stortlocatie)	Biogasproductie in een kort tijdsbestek
	Minder afval voor de eindbehandeling
	Minder afval voor behandeling in BIOCEL reactors
	Minder afval voor behandeling in de Bioreactorstort
Voordelen afvalbehandeling in een Bioreactorstort in vergelijking met een Standaard stort	Minder percolatiewater dat uiteindelijk moet worden gezuiverd op de locatie.
	Versnelde stabilisatie in een korter tijdsbestek (10 jaar)
	Efficiënte benutting van de stortlocatie
	Vermindering nazorg bij sluiting
	Meer biogas
Cel als basisonderdeel van de Bioreactorstort • Lay-out • Management	Minder broeikasgassen
	>10 m celdiepte
	Celvulling op weekbasis
	5 jaar cel activiteit (recirculatie perculatiewater, gas opvang)
	Na 5 jaar wordt de cel partieel actief (geen recirculatie van het percolatiewater, wel opvang stortgas)
Stortgas (biogas)	Na 10 jaar wordt de cel volledig gesloten (geen opvang percolatiewater, geen gas opvang)
	Gasopvang
Percolatiewater management	Vermijding emissie van broeikasgassen
	Inzameling percolatiewater
	Vertikale bronnen voor recirculatie van het percolatiewater
	In-situ behandeling

De vermelde aspecten hebben betrekking op het anaërobe bedrijven van de Biogasstort met als belangrijk voordeel de productie van biogas als gevolg van de anaërobe afbraak van organisch materiaal dat het hoofdbestanddeel is van het stedelijk afval in Oost-Afrika. Een ander belangrijk aspect is de introductie van het BIOCEL-systeem als voorbehandelingssysteem voor het stedelijk afval. Het BIOCEL systeem is een bewezen technologie met het vermogen om in korte tijd (ongeveer 20 dagen) grote hoeveelheden biogas te produceren uit de snel afbreekbare organische fractie in het stedelijk afval. Daarbij wordt ook een forse reductie van de hoeveelheid afval verkregen. Relevante aspecten bij de toepassing van een celsysteem bij een stort zijn o.a. de vultijd van de cel, de duur van de actieve periode, hoogte, opvang en recirculatie van percolatiewater en gasopvangsysteem. De karakteristieken in de tabel verwijzen ten dele ook naar verdiensten die kunnen worden verkregen bij toepassing van een Bioreactorstort. In vergelijking met de storten zoals die momenteel in Oost-Afrika worden toegepast zijn de verdiensten van een Bioreactorstort een versnelde stabilisatie van het afval, efficiënter gebruik van de stortruimte, een grotere hoeveelheid biogas verkregen in een kortere tijdsperiode, vermijding van emissie van broeikasgassen, een betere controle van stank, lagere kosten voor behandeling van het te lozen percolatiewater en lagere kosten van beheer van de stort nadat die is gesloten.

Hoofdstuk 4 gaat in op de productie en kwaliteit in samenhang met de recirculatie van percolatiewater en op mogelijke technieken voor de zuivering van percolatiewater of ander afvalwater uit systemen waarin Bioreactorstortplaatsen worden gebruikt. De productie van percolatiewater is in hoge mate afhankelijk van het initiële vochtgehalte van het afval. De kwaliteit hangt af van een reeks gelijktijdige fysische, chemische en biologische processen in het afval welke resulteren in het vrijkomen van gesuspendeerde en opgeloste stoffen in de waterfase en de productie van gassen. Vanwege de inhomogene samenstelling van afval in stortplaatsen en de combinatie van deze processen kunnen de concentraties van CZV, BZV en ammonia in het percolatiewater in de tijd sterk variëren met een dalende trend naarmate het afval stabiliseert. De recirculatie van percolatiewater heeft tot doel water en hoge concentraties microorganismen door het afval ter verspreiden teneinde het stabilisatieproces te versnellen. Het hoofdstuk poogt optimale recirculatiesnelheden vast te stellen. Oververzadiging van afval met water kan leiden tot verzuring en remming van de methaanvorming. Dit verschijnsel kan gebruikt worden om gedurende een zekere tijd een tweetrapsproces te induceren. Dit kan de vorm aannemen van een Bioreactorstort gevolgd door een UASB reactor. Dit tweetrapsproces is een van de systeemopties in hoofdstuk 7.
Het vrijkomende percolatiewater moet worden gezuiverd om te voldoen aan de normen voor lozing op oppervlaktewater of op het riool. In dat opzicht moet in ogenschouw worden genomen dat Tanzania een ontwikkelingsland is, een tropisch klimaat heeft met hoge temperatuur en gemiddelde regenval, de financiële middelen van het land beperkt zijn en ook de expertise en de noodzakelijke capaciteit om geavanceerde zuiveringssystemen te exploiteren ontbreekt. Verder moet worden geconstateerd, dat de belangrijkste vervuiling in het afvalwater en percolatiewater van een Bioreactorstort bestaat uit een relatief hoge concentratie van niet of moeilijk biologisch afbreekbare CZV en een hoge concentratie ammoniak. Het percolatiewater wordt verder gekenmerkt door lage concentraties van BZV en van vluchtige vetzuren. Op basis van deze gegevens worden met name mechanisch beluchte aërobe zuivering, vijversystemen en helofytenfilters als geschikte technieken voor de komende 10 à 15 jaar beschouwd. Er worden op grond van het voorafgaande vier mogelijkheden voor een zuiveringsproces naar voren gebracht:

- Actief slib proces in combinatie met een helofytenfilter als nabehandeling;
- Facultatief vijversysteem met een helofytenfilter als nabehandeling;
- Verdamping;
- Gecombineerd biologische en fysisch-chemische zuivering.

De eerste drie systemen voldoen aan de typische condities en beperkingen in Tanzania en in Oost-Afrika in het algemeen. De vierde methode is een meer geavanceerde zuivering gebaseerd op de toepassing van batchreactor technologie - coagulatie/flocculatie - Fenton oxidatie - aërobe biologische nazuivering. Deze vierde methode wordt, gezien de eerder genoemde condities en beperkingen op dit moment niet haalbaar geacht, maar mogelijk wel in de toekomst.

In hoofdstuk 5 wordt een overzicht gegeven van de modellen die kunnen worden toegepast voor het berekenen van de degradatie van afval en de productie van biogas in een Bioreactorstort. De nadruk van dit hoofdstuk ligt op de modellering van de productie en productiesnelheid van stortgas uit een Bioreactorstort. In eerste instantie wordt een uitgebreid overzicht gegeven van de modellen die in de literatuur worden vermeld voor de kwantificering van de productie van biogas en methaan. Deze modellen worden kritisch geëvalueerd. Alle modellen die de productie van methaan of biogas weergeven zijn gebaseerd op een eerste orde reactiesnelheid in de totale hoeveelheid afval, de totale hoeveelheid organisch materiaal in het afval, of de totale hoeveelheid organische koolstof in het afval. Deze modellen ondersteunen de berekening van de productiesnelheid van stortgas en de berekening van het totale potentieel aan biogas of methaan dat in het afval aanwezig is, maar niet alle modellen gebruiken dezelfde invoergegevens. In dit proefschrift wordt een verbeterd, gemodificeerd model voor de berekening de biogasproductie uit afval gepresenteerd. Dit model is ontwikkeld voor de berekening van de biogasproductie van een compartiment gestort afval als functie van de tijd. Hierbij is aangenomen dat alle afval in het compartiment dezelfde leeftijd heeft. Maar het model kan ook worden toegepast voor de berekening over een bepaald specifiek tijdsverloop van de totale biogas productie uit een aantal compartimenten met verschillende leeftijd van het gestorte afval**.** Het gemodificeerde model is zeer inzichtelijk en wordt toegepast in hoofdstuk 6 van dit proefschrift.

In hoofdstuk 6 wordt het experimenteel onderzoek aan een Bioreactorstort op pilotplant schaal beschreven. Dit pilotplantonderzoek werd uitgevoerd in Dar es Salaam City, Tanzania, Oost-Afrika. Het doel van dit onderzoek was meer te weten te komen over het effect van recirculatie van percolatiewater op de afbraak en verzuring van het afval, de stortgas productie en de in-situ zuivering van het percolatiewater. Om dit doel te bereiken werden de volgende activiteiten uitgevoerd: (1) Studie van de variatie in de karakteristieken van het uittredende percolatiewater als een indicator voor de afvalstabilisatie, (2) Effect van de recirculatie van het percolatiewater op de afbraak van de CZV in dit percolatiewater, (3) Evaluatie van de snelheid van biogasproductie en de samenstelling van het biogas, (4) Monitoring van de inklinking/compactering van het afval als gevolg van de biodegradatie van het afval. De pilotplant was opgebouwd uit twee reactoren: reactor R1, waarbij geen recirculatie van percolatiewater werd toegepast en reactor R2 waarbij wel recirculatie van percolatiewater werd toegepast. Reactor R1 werd bedreven als controle reactor, reactor R2 werd bedreven als een gesimuleerde Bioreactorstort. De reactoren werden gelijktijdig gevuld, elk met ongeveer 2,3 ton afval met een vochtgehalte van ongeveer 60%. Tijdens de onderzoekperiode van 52 weken werd reactor R1 bedreven als een doorstroomreactor. Bij reactor R2 werd gedurende de eerste periode recirculatie van percolatiewater toegepast. Het percolatiewater werd daarbij aan de bovenzijde van de reactor toegevoerd. Tijdens de tweede periode werd het aan de onderzijde opgevangen percolatiewater eerst in een UASB (Up-flow Anaerobic Sludge Blanket) reactor gezuiverd alvorens het weer aan toe te dienen aan de top van reactor R2. Deze UASB reactor was opgebouwd uit PVC, had een inhoud van 15,7 l, een hoogte van 2 m een diameter van 0,1 m, een hydraulische verblijftijd van 1,15 dagen en was gevuld met 6,75 l anaërobe slib. Het slib was afkomstig van een bestaande UASB reactor en had een leeftijd van meer dan 5 jaar en een biologische activiteit van ongeveer 0.17 g CZV/g VSS/dag. De belangrijkste resultaten van dit onderzoek bevestigen de haalbaarheid van een Bioreactorstort voor de behandeling van afval dat karakteristiek is voor Oost-Afrika. De techniek leent zich goed voor een versnelde stabilisatie van afval, dat rijk is aan organische stof, leverteen hoge stortgasproductie vanwege het hoge gehalte aan organische stof en brengt ook een zekere mate van zuivering van het percolatiewater teweeg.

Naast bovengenoemde resultaten kan nog een aantal specifieke conclusies worden getrokken: gecontroleerde verzuring van het percolatiewater is gedurende een langere periode mogelijk. In de praktijk betekent dit dat bij dit tweetrapsproces geen of weinig biogas wordt geproduceerd in de Bioreactorstort zelf, zodat er geen of weinig verlies van methaan uit de stort plaats vindt. Dit betekent ook, dat het tweetrapsproces in zijn geheel gezien resulteert in minder verlies aan stortgas. Uit het onderzoek volgt verder dat wel speciale aandacht moet worden besteed aan de verwijdering van nutriënten (N en P) uit het percolatiewater, omdat in het toegepaste reactor systeem geen verwijdering van deze componenten plaats vindt.

In hoofdstuk 7 worden vier innovatieve concepten (Systeemopties) gepresenteerd die aangepast zijn aan de Oost-Afrikaanse situatie en die gebaseerd zijn op geavanceerde bestaande kennis en op de resultaten van onderzoek op pilotplantschaal. In dit hoofdstuk wordt tevens een schematisch ontwerp en een schematische weergave van de werking van deze opties weergegeven. De ontwikkelde concepten omvatten tevens de terugwinning van materialen en het gebruik van overslagstations. Op deze overslagstations wordt ook de niet-biologisch afbreekbare fractie uit het gemengde afval verwijderd voordat dit naar de Bioreactorstort gaat of voor behandeling naar een aparte anaërobe bioreactor, BIOCEL, wordt afgevoerd. De concepten omvatten naast de Bioreactorstort en de BIOCEL ook andere reactoren ter ondersteuning van het afbraakproces van afval en afvalcomponenten, zoals het opslag- en recirculatie systeem van het percolatiewater, de voorbehandeling en nabehandeling van het percolatiewater en het stortgasopvangsysteem. In hoofdstuk 3 en hoofdstuk 4 zijn deze systemen uitvoerig beschreven en geëvalueerd. In hoofdstuk 5 zijn de modellen uitgewerkt waarmee de biogasproductie kan worden berekend. De ontwikkeling van deze innovatieve concepten is uiteindelijk verkregen middels: a) Empirisch onderzoek, uitgevoerd in Oost-Afrika-Tanzania, en gericht op het inventariseren en evalueren van de bestaande managementpraktijk betreffende stedelijk afval en de karakterisering van de hoeveelheid en samenstelling van het ingezamelde stedelijk afval (hoofdstuk 2); b) Een uitgebreide literatuurstudie betreffende Bioreactorstorten en behandelingssystemen voor percolatiewater (hoofdstuk 3 en hoofdstuk 4); c) Modellering van de stortgasproductie (hoofdstuk 5); d) Lokaal uitgevoerd pilotplantonderzoek (hoofdstuk 6). De verschillende Systeemopties zijn o.a. gebaseerd op verschillen in de hoeveelheid en samenstelling van het afval dat uiteindelijk gestort moet worden, de productie en behandeling van percolatiewater en de plaats waar het stortgas of biogas wordt geproduceerd. Deze systeemopties zijn:

- Systeemoptie 1: Standaard Bioreactorstort;
- Systeemoptie 2: Standaard Bioreactorstort waarbij een deel van de stortgas- (biogas-) productie plaats vindt in een UASB reactor op de stortplaats;
- Systeemoptie 3: Twee-traps behandeling, eerst in een centrale BIOCEL op de stort gevolgd door een verdere behandeling in de Bioreactorstort;
- Systeemoptie 4: Twee-traps behandeling bestaande uit decentrale voorbehandeling in een BIOCEL op de overslagstations en vervolgbehandeling in de Standaard Bioreactorstort.

Voor een aantal standaardcondities en aannames werd voor iedere systeemoptie de hoeveelheid stortgas berekend alsmede de emissie van methaan, de grootte van de stort en de hoeveelheid te zuiveren percolatiewater. Op deze wijze konden de diverse systeemopties met elkaar worden vergeleken. Het afval in Oost-Afrika wordt gekarakteriseerd door een watergehalte van ca. 60%, wat erg hoog is en aanzienlijk boven de veldcapaciteit van het afval ligt. Dit was een reden om voor de vier systeemopties na te gaan wat de mogelijke consequenties kunnen zijn van een gedeeltelijk drainering van dit afval voordat het wordt toegevoerd aan de BIOCEL of gestort wordt op de Bioreactorstort. Verder werd een gevoeligheidsanalyse uitgevoerd met betrekking tot een aantal aannames. Uit deze gevoeligheidsanalyse blijkt, dat de hoeveelheid stortgas die wordt opgevangen alsmede de emissie van broeikasgassen niet erg gevoelig zijn voor kleine veranderingen in de efficiëntie van het opvangsysteem voor stortgas. Procesvoering welke gericht is op verzuring van het percolatiewater dat onttrokken wordt aan relatief vers gestort afval, vereist weliswaar een meer intensieve procescontrole maar is wel erg zinvol bij een relatief lage opvangefficiëntie van het geproduceerde stortgas. Bij een relatief laag opvangpercentage van het biogas heeft ook de toepassing van een BIOCEL een groot effect op het ruimtebeslag van de stort en op de emissie van broeikasgassen. Meer nauwkeurige waarden van procesparameters zijn vereist voor meer nauwkeurige en betrouwbare berekeningen, speciaal wat betreft het biogas potentieel van systemen waarin een BIOCEL is opgenomen, het effect van de oxidatieëfficiëntie van de toplaag, het opvangpercentage van het geproduceerde stortgas, de emissie van broeikasgassen en de periode dat verzuring van het percolatiewater kan worden gehandhaafd. Dergelijke nauwkeuriger waarden kunnen middels experimenteel onderzoek verkregen worden.

De kosten van alle vier systeemopties zijn hoger dan die van een conventionele stort. Dit is het gevolg van de toepassing van een opvang- en recirculatiesysteem voor het percolatiewater en de toepassing van een gasopvangsysteem. De hogere investeringskosten kunnen echter worden gecompenseerd, omdat meer stortgas en biogas beschikbaar komt voor elektriciteitsproductie en inkomsten via het Clean Development Mechanism (CDM) kunnen worden verkregen vanwege vermeden emissies van broeikasgassen. Met name bij systemen waarin een BIOCEL is opgenomen is dit het geval. Het ontwikkelde berekeningsmodel kan ook gemakkelijk worden toegepast voor procesvariabelen die sterk afwijkend zijn van de standaard procesvariabelen. Tezamen met de vier systeemopties vormt het berekeningsmodel een zeer bruikbaar instrument om beslissingen te onderbouwen of te nemen inzake de keuze van een afvalmanagementsysteem in Oost-Afrikaanse landen.

Hoofdstuk 8 omvat een algemene discussie van de onderzoekresultaten en een samenvatting van de conclusies van de voorafgaande zeven hoofstukken van dit proefschrift. Het hoofddoel van dit proefschrift was om een Bioreactorstort voor stedelijk afval te ontwerpen en te beschrijven die geschikt is voor toepassing in Oost-Afrikaanse steden. In dit hoofdstuk vindt een kritische terugblik plaats op de uitwerking en evaluatie van de verschillende opties van een Bioreactorstort. Daarbij worden de volgende algemene eindconclusies getrokken: (1). Stedelijk afval in Oost-Afrika wordt gekenmerkt door een zeer hoog gehalte aan organisch materiaal en een watergehalte van meer dan 60%. Storten van afval is een wezenlijk element van een geïntegreerde afvalmanagement strategie. Het is te verwachten dat een meer geavanceerde en moderne vorm van een storten, zoals de Bioreactorstort op korte en middellange termijn belangrijk zal worden voor de behandeling van stedelijk afval in Oost-Afrika. (2). Er werden vier innovatieve modificaties (systeem opties) van een Bioreactorstort geïdentificeerd, uitgewerkt en geëvalueerd op basis van literatuurinformatie over de constructie en werking van Bioreactorstorten in hooggeïndustrialiseerde Westerse landen, de specifieke kenmerken van stedelijk afval in Oost-Afrika, de resultaten van experimenteel onderzoek verkregen met behulp van een pilot installatie en een aantal bureaustudies betreffende de biologische omzetting, het modelleren van de biologische omzettingssnelheid en de biogasproductie uit stedelijk afval. (3) Deze vier opties werden met behulp van een semi-mathematisch berekeningsmodel geëvalueerd wat betreft investering, operationele kosten, landbehoefte, kosten van en besparingen op behandeling van percolatiewater, stortgasproductie, stortgasopvang, opbrengst van het stortgas, hergebruik van stortruimte en vermijding van emissie van broeikasgassen. Vergeleken met een simpele gecontroleerde stort voor stedelijk afval zoals die in Oost-Afrika momenteel wordt toegepast, bieden de vier modificaties van de Bioreactorstort grote voordelen wat betreft ruimtebeslag, hoeveelheid stortgas die wordt gewonnen en de emissie van broeikasgassen.

REFERENCES

Abbas, A. A., JingSong Guo, LiuZhi Ping, PanYing Ya, and W. S. Al-Rekabi. (2009). Review on landfill leachate treatments. *Journal of Applied Sciences Research* May:534-545.

Ağdağ, O. N., and D. T. Sponza. (2005). Effect of alkalinity on the performance of a simulated landfill bioreactor digesting organic solid wastes. *Chemosphere* 59 (6):871-9.

Ağdağ, O. N., and D. T. Sponza. (2007). Co-digestion of mixed industrial sludge with municipal solid wastes in anaerobic simulated landfilling bioreactors. *Journal of Hazardous Materials* 140 (1-2):75-85.

Ahn, Won-Young, Moon-Sun Kang, Seong-Keun Yim, and Kwang-Ho Choi. (2002). Advanced landfill leachate treatment using an integrated membrane process. *Desalination* 149 (1–3):109-114.

Al-Khatib, I. A., H. A. Arafat, T. Basheer, H. Shawahneh, A. Salahat, J. Eid, and W. Ali. (2007). Trends and problems of solid waste management in developing countries: A case study in seven Palestinian districts. *Waste Management* 27 (12):1910-1919.

Al-Yousfi, A. B. (1992). Modeling of leachate and gas production and composition at sanitary landfills, University of Pittsburgh, Pittsburgh, PA.

Allen, A. (2001). Containment landfills: the myth of sustainability. *Engineering Geology* 60 (1-4):3-19.

Alvarez-Vazquez, H., B. Jefferson, and S. J. Judd. (2004). Membrane bioreactors vs conventional biological treatment of landfill leachate: a brief review. *Journal of Chemical Technology and Biotechnology* 79 (10):1043-1049.

APHA. (2005). *Standard methods: For the examination of water and wastewater*. Edited by A. D. Eaton, L. S. Clesceri, E. W. Rice and A. E. Greenberg. 21st ed. Washington D.C.: American Public Health Association, American Water Works Association, Water Environment Federation.

Arthur, J. (1983). Notes on the design and operation of waste stabilization ponds in Warm Climates of Developing Countries. In *World Bank Technical Paper nr 6*. Washington.

Bae, J.H., K.W. Cho, B.S. Bum, S.J. Lee, and B.H. Yoon. (1998). Effects of leachate recycle and anaerobic digester sludge recycle on the methane production from solid waste. *Water Science Technology* 38:159-168.

Baldwin, T. D., J. Stinson, and R. K. Ham. (1998). Decomposition of specific materials buried within sanitary landfills. *Journal of Environmental Engineering (New York)* 124 (12):1193-1202.

Bareither, C. A., C. H. Benson, M. A. Barlaz, T. B. Edil, and T. M. Tolaymat. (2010). Performance of North American bioreactor landfills. I: Leachate hydrology and waste settlement. *Journal of Environmental Engineering-Asce* 136 (8):824-838.

Barlaz, B. A., R. K. Ham, and D. M. Schaefer. (1989). Mass-balance analysis of anaerobically decomposed refuse. *Journal of Environmental Engineering-Asce* 115 (6):1088-1102.

Barlaz, M. A., R. K. Ham, and D. M. Schaefer. (1990). Methane production from municipal refuse: A review of enhancement techniques and microbial dynamics. *CRC Critical Reviews in Environmental Control* 19 (6):557-584.

Barlaz, M., and D. Reinhart. (2004). Bioreactor landfills: Progress continues. *Waste Management* 23 (7):557-559.

Benbelkacem, H., R. Bayard, A. Abdelhay, Y. Zhang, and R. Gourdon. (2010). Effect of leachate injection modes on municipal solid waste degradation in anaerobic bioreactor. *Bioresour Technol* 101 (14):5206-12.

Benson, C. H., M. A. Barlaz, D. T. Lane, and J. M. Rawe. (2007). Practice review of five bioreactor/recirculation landfills. *Waste Management* 27 (1):13-29.

Berge, N. D., D. R. Reinhart, J. D. Dietz, and T. Townsend. (2007). The impact of temperature and gas-phase oxygen on kinetics of in situ ammonia removal in bioreactor landfill leachate. *Water Research* 41 (9):1907-1914.

Berge, N. D., D. R. Reinhart, and T. G. Townsend. (2005). The fate of nitrogen in bioreactor landfills. *Critical Reviews in Environmental Science and Technology* 35 (4):365-399.

Berge, Nicole D., Debra R. Reinhart, and Eyad S. Batarseh. (2009). An assessment of bioreactor landfill costs and benefits. *Waste Management* 29 (5):1558-1567.

Bozkurt, S., L. Moreno, and I. Neretnieks. (2000). Long-term processes in waste deposits. *Sci Total Environ* 250 (1-3):101-21.

Burton, S. A. Q. , and I. A. Watson-Craik. (1998). Ammonia and nitrogen fluxes in landfill sites: applicability to sustainable landfilling *Waste management and research* 16 (1):41-53.

Buswell, E. G. , and S. L. Neave. (1930). Laboratory studies of sludge digestion [Article]. *Illinois Division of State Water Survey.* Vol. Bulletin no. 30.

Capela, I. F., C. Azeiteiro, L. Arroja, and A. C. Duarte. (1999). Effects of pre-treatment (composting) on the anaerobic disgestion of primary sludges from a bleached kraft pulp mill. In *II International Symposium on Anaerobic Digestion of Solid Waste*. Barcelona 15-17 June.

Cecen, F., and O. Aktas. (2001). Effect of PAC addition in combined treatment of landfill leachate and domestic wastewater in semi-continuously fed batch and continuous-flow reactors. *Water Sa* 27 (2):177-188.

Cecen, F., and O. Aktas. (2004). Aerobic co-treatment of landfill leachate with domestic wastewater. *Environmental Engineering Science* 21 (3):303-312.

Census. (2003). Population and Housing. In *Census General report* edited by D. o. P. C. a. S. Statistics: National Bureau of Statistics, United Republic of Tanzania.

Chaggu, E. J. (2004). Sustainable environmental protection using modified pit-latrines, Environmental Technology, Wageningen University, Wageningen.

Chen, Ying-xu, Song-wei Wu, Wei-xiang Wu, Hua Sun, and Ying Ding. (2009). Denitrification capacity of bioreactors filled with refuse at different landfill ages. *Journal of Hazardous Materials* 172 (1):159-165.

Chong, T. L., Y. Matsufuji, and M. N. Hassan. (2005). Implementation of the semi-aerobic landfill system (Fukuoka method) in developing countries: A Malaysia cost analysis. *Waste Management* 25 (7):702-711.

Christensen, T. H., and P. Kjeldsen. (1989). *Basic biochemical processes in landfills. In Sanitary landfilling, Process, Technology and Environmental Impact*. Toronto: Academic Press Ltd.

Chugh, S., W. Clarke, P. Pullammanappallil, and V. Rudolph. (1998). Effect of recirculated leachate volume on MSW degradation. *Waste Management Research* 16 (6):564-573.

Cointreau Levine, S. (1994). *Private sector participation in municipal solid waste services in developing countries*. Vol. 1, *The formal sector*. Washington, DC: UNDP / UNCHS / World Bank - Urban Management Programme.

Cooney, M., N. Maynard, C. Cannizzaro, and J. Benemann. (2007). Two-phase anaerobic digestion for production of hydrogen-methane mixtures. *Bioresour Technol* 98 (14):2641-51.

DCC. (2004). Dar es Salaam City Profile, City Director, Dar es Salaam.

de Mes, T. Z. D., A. J. M. Stams, J. H. Reith, and G. Zeeman. (2003). Methane production by anaerobic digestion of wastewater and solid wastes. In *Bio-methane & Bio-hydrogen: Status and perspectives of biological methane and hydrogen production*, edited by J. H. Reith, R. H. Wijffels and H. Barten. The Netherlands: Dutch Biological Hydrogen Foundation, c/o Energy research Centre

de Velasquez, M. T. Orta, I. Monje-Ramirez, and I. Yañez Noguez. (2011). Saline landfill leachate disposal in facultative lagoons for wastewater treatment. *Environmental technology* 33 (2):247-255.

Diamadopoulos, E., P. Samaras, X. Dabou, and G. P. Sakellaropoulos. (1997). Combined treatment of landfill leachate and domestic sewage in a sequencing batch reactor. *Water Science and Technology* 36 (2-3):61-68.

Dinamarca, S., G. Aroca, R. Chamy, and L. Guerrero. (2003). The influence of pH in the hydrolytic stage of anaerobic digestion of the organic fraction of urban solid waste. *Water Sci Technol* 48 (6):249-54.

EAWAG, Sandec, ed. (2006). *Decentralised composting for cities of low- and middle-income countries*, *A Users's manual*.

El-Fadel, M., E. Bou-Zeid, W. Chahine, and B. Alayli. (2002). Temporal variation of leachate quality from pre-sorted and baled municipal solid waste with high organic and moisture content. *Waste Management* 22 (3):269-282.

El-Fadel, M., A. N. Findikakis, and J. O. Leckie. (1997). Gas simulation models for solid waste landfills. *Critical Reviews in Environmental Science and Technology* 27 (3):237 - 283.

El-Fadel, M., and R. Khoury. (2000). Modeling settlement in MSW landfills: a critical review. *Critical Reviews in Environmental Science and Technology* 30 (3):327-361.

Elagroudy, S. A., M. H. Abdel-Razik, M. A. Warith, and F. H. Ghobrial. (2008). Waste settlement in bioreactor landfill models. *Waste Management* 28 (11):2366-2374.

EMA. (2004). Environmental Management Act. United Republic of Tanzania.

Erses, S. A., and T. T. Onay. (2003). In situ heavy metal attenuation in landfills under methanogenic conditions. *J. Hazard Mater* (99):159-75.

Frascari, D., F. Bronzini, G. Giordano, G. Tedioli, and A. Nocentini. (2004). Long-term characterization, lagoon treatment and migration potential of landfill leachate: a case study in an active Italian landfill. *Chemosphere* 54 (3):335-343.

Gawande, N. A., D. R. Reinhart, and G. T. Yeh. (2010a). Modeling microbiological and chemical processes in municipal solid waste bioreactor, part I: development of a three-phase numerical model BIOKEMOD-3P. *Waste Management* 30 (2):202-210.

Gawande, N. A., D. R. Reinhart, and G. T. Yeh. (2010b). Modeling microbiological and chemical processes in municipal solid waste bioreactor, part II: application of numerical model BIOKEMOD-3P. *Waste Management* 30 (2):211-218.

Gholamifard, S., R. Eymard, and C. Duquennoi. (2008). Modeling anaerobic bioreactor landfills in methanogenic phase: long term and short term behaviors. *Water Research (Oxford)* 42 (20):5061-5071.

Gregory, R. G, A. G Gillet, and D. Bradley. (2003a). Methane emissions from landfill sites in the UK. In *LQM Report 443/1*, *January 2003*.

Gregory, R. G., M. G. Attenborough, C. D. Hall, and C. Deed. (2003b). The validation and development of an integrated landfill gas risk assessment model GasSim. Paper read at Sardinia 2003, at Cagliari, Italy.

Hansson, G. (1982). End Product Inhibition in Methane Fermentations. *Process Biochemistry* 17 (6):45-49.

Hao, Yong-Jun, Wei-Xiang Wu, Song-Wei Wu, Hua Sun, and Ying-Xu Chen. (2008). Municipal solid waste decomposition under oversaturated condition in comparison with leachate recirculation. *Process Biochemistry* 43 (1):108-112.

Haydar, M. M., and M. V. Khire. (2005). Leachate recirculation using horizontal trenches in bioreactor landfills. *Journal of Geotechnical and Geoenvironmental Engineering* 131 (7):837-847.

He, Pin-Jing, Li-Ming Shao, Xian Qu, Guo-Jian Li, and Duu-Jong Lee. (2005). Effects of feed solutions on refuse hydrolysis and landfill leachate characteristics. *Chemosphere* 59 (6):837-844.

He, Ruo, Xin-wen Liu, Zhi-jian Zhang, and Dong-sheng Shen. (2007). Characteristics of the bioreactor landfill system using an anaerobic-aerobic process for nitrogen removal. *Bioresource Technology* 98 (13):2526-2532.

Heavey, M. (2003). Low-cost treatment of landfill leachate using peat. *Waste Management* 23 (5):447-454.

Henry, Rotich K. , Yongsheng, Zhao. Jun, Dong. (2006). Municipal solid waste management challenges in developing countries – Kenyan case study. *Waste Management* 26:92-100.

Hesagawa, S., and K. Katsura. (1999). Solubilization of organic sludge by thermophilic aerobi bacteria for asa pretreatment for anaerobic digestion. In *II International Symposium on Anaerobic Digestion of Solid Waste*. Barcelona June 15-17

Hettiarachchi, C. H. (2005). Mechanics of biocell landfill settlements: New Jersey Institute of Technology. Dept. of Civil and Environmental Engineering.,.

Hettiarachchi, H., J. Meegoda, and P. Hettiaratchi. (2009). Effects of gas and moisture on modeling of bioreactor landfill settlement. *Waste Manag* 29 (3):1018-25.

Hristovski, K., L. Olson, N. Hild, D. Peterson, and S. Burge. (2007). The municipal solid waste system and solid waste characterization at the municipality of Veles, Macedonia. *Waste Management* 27 (11):1680-1689.

IPCC. (2001). Third assessment report. In *Emissions of greenhouse gases in the United States 2003*. U.S: IPCC and U.S. Energy Information Administration (EIA).

IPCC. (2006). Guidelines for national greenhouse gas inventories. In *Biological treatment of solid waste*. IGES, Japan: Intergovernmental Panel on Climate Change.

ITRC. (2005). Characterization, design, construction, and monitoring of bioreactor landfills. ALT-3. In *(ITRC) Interstate Technology & Regulatory Council, Alternative Landfill Technologies Team*. Washington, DC.

Jain, P., J. Powell, T.G. Townsend, and D.R. Reinhart. (2005). Moisture addition at a bioreactor landfill using vertical wells. *Journal of Environmental Engineering Science*.

Jiang, J., G. Yang, Z. Deng, Y. Huang, Z. Huang, X. Feng, S. Zhou, and C. Zhang. (2007). Pilot-scale experiment on anaerobic bioreactor landfills in China. *Waste Management* 27 (7):893-901.

JICA. (1996). The study on solid waste management for Dar es Salaam city. Dar es Salaam: Unpublished work.

Kaseva, M. E., and S. E. Mbuligwe. (2000). Ramifications of solid waste disposal site relocation in urban areas of developing countries: A case study in Tanzania. *Resources, Conservation and Recycling* 28 (1-2):147-161.

Kaseva, M. E., and S. E. Mbuligwe. (2005). Appraisal of solid waste collection following private sector involvement in Dar es Salaam city, Tanzania. *Habitat International* 29 (2):353-366.

Kasozi, A. , and H. von Blottnitz. (2010). Solid waste management in Nairobi: A situation analysis. In *Technical document accompanying the integrated solid waste management plan*: For the City Council of Nairobi on contract for the United Nations Environment Programme.

Kassim, S. M., and M. Ali. (2006). Solid waste collection by the private sector: Households' perspective--Findings from a study in Dar es Salaam city, Tanzania. *Habitat International* 30 (4):769.

KCC. (2006). Kampala solid waste management strategy. edited by K. C. Council. Kampala: MLG, Republic of Uganda.

Khire, M. V., and M. M. Haydar. (2007). Leachate recirculation in bioreactor landfills using geocomposite drainage material. *Journal of Geotechnical and Geoenvironmental Engineering* 133 (2):166-174.

Khire, M. V., and M. Mukherjee. (2007). Leachate injection using vertical wells in bioreactor landfills. *Waste Management* 27 (9):1233-1247.

Kiely, G. (1997). *Envirnomental Engineering*. London: McGraw Hill.

Kim, Dong-Jin, Dong-Ig Lee, and Jürg Keller. (2006). Effect of temperature and free ammonia on nitrification and nitrite accumulation in landfill leachate and analysis of its nitrifying bacterial community by FISH. *Bioresource Technology* 97 (3):459-468.

Kim, I. S., D. H. Kim, and S. H. Hyun. (2000). Effect of particle size and sodium ion concentration on anaerobic thermophilic food waste digestion. *Water Sci Technol* 41 (3):67-73.

Kim, J., and F. G. Pohland. (2003). Process enhancement in anaerobic bioreactor landfills. *Water Science and Technology* 48 (4):29-36.

Kjeldsen, Peter, Morton A. Barlaz, Alix P. Rooker, Anders Baun, Anna Ledin, and Thomas H. Christensen. (2002a). Present and Long-Term Composition of MSW Landfill Leachate: A Review. *Critical Reviews in Environmental Science and Technology,* 32 (4):297-336.

Kjeldsen, Peter, Morton A. Barlaz, Alix P. Rooker, Anders Baun, Anna Ledin, and Thomas H. Christensen. (2002b). Present and Long-Term Composition of MSW Landfill Leachate: A Review. *Critical Reviews in Environmental Science and Technology,* 32 (4):297-336.

Kulikowska, D., and E. Klimiuk. (2008). The effect of landfill age on municipal leachate composition. *Bioresource Technology* 99 (13):5981-5985.

Lema, J. M., R. Mendez, and R. Blazquez. (1988). Characteristics of landfill leachates and alternatives for their treatment: A review. *Water, Air, & Soil Pollution* 40 (3):223-250.

Lens, P. , B. Hamelers, H. Hoitink, and W. Bidlingmaier, eds. (2004). *Integrated Environmental Technology Series, Resource recovery and reuse in organic solid waste management*. London, UK: IWA Publishing.

Li, Huo-sheng, Shao-qi Zhou, Yan-bo Sun, Ping Feng, and Jing-da Li. (2009). Advanced treatment of landfill leachate by a new combination process in a full-scale plant. *Journal of Hazardous Materials* 172:408–415.

Long, Yu-Yang, Li-Fang Hu, Cheng-Ran Fang, Ruo He, and Dong-Sheng Shen. (2009a). Releasing behavior of zinc in recirculated bioreactor landfill. *Science of the Total Environment* 407 (13):4110-4116.

Long, Yu-Yang, Li-Fang Hu, Chen-Jing Jiang, Cheng-Ran Fang, Feng-Ping Wang, and Dong-Sheng Shen. (2009b). Releasing behavior of copper in recirculated bioreactor landfill. *Bioresource Technology* 100 (8):2419-2424.

Lozecznik, S., R. Sparling, J. A. Oleszkiewicz, S. Clark, and J. F. VanGulck. (2010). Leachate treatment before injection into a bioreactor landfill: Clogging potential

reduction and benefits of using methanogenesis. *Waste Management* 30 (11):2030-2036.
Machado, S. L., M. F. Carvalho, J. P. Gourc, O. M. Vilar, and J. C. do Nascimento. (2009). Methane generation in tropical landfills: Simplified methods and field results. *Waste Management* 29 (1):153-61.
Manfredi, S., D. Tonini, T. H. Christensen, and H. Scharff. (2009). Landfilling of waste: accounting of greenhouse gases and global warming contributions. (Special Issue: Fundamentals in greenhouse gas accounting: concepts and mechanisms.). *Waste Management & Research* 27 (8):825-836.
Mata-Alvarez, J. . (2002). *Biomethanization of the organic fraction of municipal solid wastes*. London, UK: IWA Publishing.
Mata-Alvarez, J., S. Macé, and P. Llabrés. (2000). Anaerobic digestion of organic solid wastes. An overview of research achievements and perspectives. *Bioresource Technology* 74 (1):3-16.
Mbuligwe, S. E. , and G. R. Kassenga. (2007). Landfill leachate pollution: Genesis, environmental implications and remediation engineering options. *The Journal of Building and Land Development* 14 (1):122-145.
Mbuligwe, S. E., G. R. Kassenga, M. E. Kaseva, and E. J. Chaggu. (2002). Potential and constraints of composting domestic solid waste in developing countries: findings from a pilot study in Dar es Salaam, Tanzania. *Resources Conservation and Recycling* 36 (1):45-59.
McBean, E. A., S. Syed-Ritchie, and F. A. Rovers. (2007). Performance results from the Tucumán solid waste bioreactor. *Waste Management* 27 (12):1783-1791.
McBean, E.A., F.A. Rovers, and G.J. Farquhar. (1995). *Solid Waste Landfill Engineering and Design*. Englewood Cliffs, New Jersey: Prentice Hall.
MCC. (2006a). Mwanza city profile. edited by C. Director. Mwanza: City Council, Tanzania.
MCC. (2006b). The preventive health services profile of Mwanza City. edited by C. H. Office: Mwanza city Council.
McCreanor, P. T. (1998). Landfill leachate recirculation systems: Mathematical modeling and validation, University of Central Florida, Orlando, FL.
McCreanor, P. T., and D. R. Reinhart. (1999). Hydrodynamic modeling of leachate recirculating landfills. *Waste Management & Research* 17:465-469.
Mehta, R., M. A. Barlaz, R. Yazdani, D. Augenstein, M. Bryars, and L. Sinderson. (2002). Refuse decomposition in the presence and absence of leachate recirculation. *Journal of Environmental Engineering* 128 (3):228-236.
Mor, S., K. Ravindra, A. De Visscher, R. P. Dahiya, and A. Chandra. (2006). Municipal solid waste characterization and its assessment for potential methane generation: A case study. *Science of the Total Environment* 371 (1–3):1-10.
Morris, J. W. F., N. C. Vasuki, J. A. Baker, and C. H. Pendleton. (2003). Findings from long-term monitoring studies at MSW landfill facilities with leachate recirculation. *Waste Management* 23 (7):653-666.
NBS. (2006). Census 2002. In *Analytical report*, edited by N. B. o. Statistics. Dar Es Salaam: Ministry of Planning, Economy and Empowerment.
Office of Auditor General. (2010). Value for money audit report on solid waste management in Kampala. Kampala: Republic of Uganda.
Onay, T. T., and F. G. Pohland. (2001). Nitrogen and sulfate attenuation in simulated landfill bioreactors. *Water Science and Technology* 44 (2-3):367-372.
Oonk, Hans (2010). Literature Review: Methane from landfills - Methods to quantify generation, oxidation and emission. Assendelft, The Netherlands: Sustainable Landfill Foundation, c/o NV Afvalzorg Holding.

Oonk, J, and A Boom. (1995). Landfill gas formation, recovery and emissions. In *NOVEM Programme from Waste and Biomass (EWAB)*. Apeldorn, Netherlands: TNO report 94-315.

Øygard, J. K., E. Gjengedal, and H. J. Mobbs. (2008). Trace element exposure in the environment from MSW landfill leachate sediments measured by a sequential extraction technique. *J. Hazard Mater* 153 (751-8).

Pacey, J., D. R. Reinhart, D. L. Hansen, G. T. Townsend, and W. H. Johnson. (1999). The bioreactor landfill: An innovation in solid waste management. Paper read at SWANA 22nd Annual LFG Symposium, at Silver springs, Maryland.

Pagilla, Krishna R., Kent C. Craney, and Wendell H. Kido. (1996). Aerobic thermophilic pretreatment of mixed sludge for pathogen reduction and Nocardia control. *Water Environment Research* 68:1093-1098.

Palmowski, L. M., and J. A. Muller. (2000). Influence of the size reduction of organic waste on their anaerobic digestion. *Water Sci Technol* 41 (3):155-62.

Parrot, L., J. Sotamenou, and B. K. Dia. (2009). Municipal solid waste management in Africa: Strategies and livelihoods in Yaoundé, Cameroon. *Waste Management* 29 (2):986-995.

Platzer, C.H.R. . (1998). Design recommendations for subsurface flow constructed wetlands for nitrification and denitrification. In *6th Int. Conf. On Wetland Systems for Water Pollution Control.* Sáo Pedro, Brazil.

Pohland, F. G. (1980). Leachate recycle as landfill management option. *Journal of the Environmental Engineering Division, ASCE* 106 (EE6):1057-1069.

Pohland, F. G., and A. B. Alyousfi. (1994). The landfill bioreactor - Principles and applications. *Abstracts of papers of the American chemical society* 208:151-ENVR.

Pohland, F. G., and S. R. Harper. (1986). Critical review and summary of leachate and gas production from landfills. In *EPA/600/2-86/073*. Cincinnati: U.S. EPA.

Pohland, F.G. (1975). Sanitary Landfill Stabilization with Leachate Recycle and Residual Treatment. In *EPA-600/2-75-043*. Cincinnati, Ohio: U.S. Environmental Protection Agency.

Powell, J. J. C., R. R. K. Turner, and I. I. J. Bateman. (2001). *Waste management and planning: Managing the Environment for Sustainable Development*. Vol. 5. Cheltham, U.K.: Edward Elgar Publishing Ltd,.

Powrie, W., and J. White. (2004). Landfill process modelling. *Waste Management* 24 (3):225-226.

Price, G. Alexander., Morton A. Barlaz, and Gary R. Hater. (2003). Nitrogen management in bioreactor landfills. *Waste Management* 23 (7):675-688.

Reed, S. C., R. W. Crites, and E. J. Middlebrooks. (1995). *Natural systems for waste management and treatment*. second ediiton ed: Mc Graw Hill

Reinhart, D, and T. Townsend. (1998). *Landfill bioreactor design and operation*. New York: Lewis Publishers.

Reinhart, D. R. , P. T. McCreanor, and T. Townsend. (2002). The bioreactor landfill: Its status and future. *Waste Management and Research* 20:172–186.

Reinhart, D. R., and A. Basel A. (1996). The impact of leachate recirculation on municipal solid waste landfill operating characteristics. *Waste Management Research* 14 (4):337-346.

Renou, S., J. G. Givaudan, S. Poulain, F. Dirassouyan, and P. Moulin. (2008). Landfill leachate treatment: Review and opportunity. *Journal of Hazardous Materials* 150 (3):468-493.

Ress, B. B., P. P. Calvert, C. A. Pettigrew, M. Barlaz, and A. (1998). Testing anaerobic biodegradability of polymers in a laboratory-scale simulated landfill. *Environmental Science & Technology* 32:821-827.

Rotich, K. H., Y. Zhao, and J. Dong. (2006). Municipal solid waste management challenges in developing countries – Kenyan study. *Waste Management* 26:92.

Salukele, F., S. Mgana, G. Szánto, J. van Buuren, and W. H. Rulkens. (2011). Municipal solid waste management diagnosis of rapidly growing cities: The case of Mwanza city, Tanzania. In *26th International Conference on Solid Waste Technology and Management*. Philadelphia, PA U.S.A.

San, I., and T. T. Onay. (2001). Impact of various leachate recirculation regimes on municipal solid waste degradation. *Journal of Hazardous Materials* 87 (1-3):259-271.

Scharff, H., and J. Jacobs. (2006). Applying guidance for methane emission estimation for landfills. *Waste Management* 26 (4):417-429.

Sharma, M., and E. McBean. (2007). A methodology for solid waste characterization based on diminishing marginal returns. *Waste Management* 27 (3):337-344.

Siegrist, H., D. Renggli, and W. Gujer. (1993). Mathematical modelling of anaerobic mesophilic sewage treatment. *Water Science and Technology* 27 (2):25-36.

Silva, A. C., M. Dezotti, and G. L. Sant'Anna. (2004). Treatment and detoxification of a sanitary landfill leachate. *Chemosphere* 55 (2):207-214.

Slack, R.J., J.R. Gronow, and N. Voulvoulis. (2005). Household hazardous waste in municipal landfills:contaminants in leachate. . *Sci Total Environ* 337 (119-37).

Sormunen, K., M. Ettala, and J. Rintala. (2008). Internal leachate quality in a municipal solid waste landfill: vertical, horizontal and temporal variation and impacts of leachate recirculation. *Journal of Hazardous Materials* 160 (2/3):601-607.

Sosnowski, P., A. Wieczorek, and S. Ledakowicz. (2003). Anaerobic co-digestion of sewage sludge and organic fraction of municipal solid wastes. *Advances in Environmental Research* 7 (3):609-616.

Spaargaren, G., P. Oosterveer, J. van Buuren, and A. P. J. Mol. (2006). Mixed modernities: Towards viable urban environmental infrastructure development in East Africa. Wageningen University.

Spagni, A., and S. Marsili-Libelli. (2009). Nitrogen removal via nitrite in a sequencing batch reactor treating sanitary landfill leachate. *Bioresource Technology* 100 (2):609-614.

Sponza, D. T., and O. N. Ağdağ. (2004). Impact of leachate recirculation and recirculation volume on stabilization of municipal solid wastes in simulated anaerobic bioreactors. *Process Biochemistry* 39 (12):2157-2165.

Sponza, D. T., and O. N. Ağdağ. (2005). Effects of shredding of wastes on the treatment of municipal solid wastes (MSWs) in simulated anaerobic recycled reactors. *Enzyme and Microbial Technology* 36 (1):25-33.

Swati, M., and K. Joseph. (2008). Settlement analysis of fresh and partially stabilised municipal solid waste in simulated controlled dumps and bioreactor landfills. *Waste Manag* 28 (8):1355-63.

Tchobanoglous, G. , and F. Kreith. (2002). *Handbook of solid waste management*. New York: McGraw-Hill.

Tchobanoglous, G., H. Theisen, and S. Vigil. (1993). *Integrated solid waste management: Engineering principles and management issues*. International ed. Boston, USA: McGraw-Hill.

ten Brummeler, E. (1993). Dry anaerobic digestion of the organic fraction of municipal solid waste. Proefschrift Wageningen, Ten Brummeler, [S.l.].

ten Brummeler, E. (2000). Full scale experience with the BIOCEL process. *Water Science and Technology* 41 (3):299-304.

ten Brummeler, E., H. C. J. M. Horbach, and I. W. Koster. (1991). Dry anaerobic batch digestion of the organic fraction of municipal solid waste. *Journal of Chemical Technology and Biotechnology* 50 (2):191-209.

Townsend, T. G. (1995). Leachate recycle at solid waste landfills using horizontal injection, University of Florida, Gainesville, FL.

Townsend, T. G., W. L. Miller, H. J. Lee, and J. F. K. Earle. (1996). Acceleration of landfill stabilization using leachate recycle. *Journal of Environmental Engineering-ASCE* 122 (4):263-268.

Townsend, T., D. Kumar, and J. Ko. (2008). Bioreactor landfill operation: A guide for development, implementation and montoring: version 1.0. Prepared for the Hinkley Center for Solid and Hazardous Waste Management, Gainesville, FL.

Tukahirwa, J. T. (2011). Civil society in urban sanitation and solid waste management: the role of NGOs and CBOs in metropolises of East Africa. Proefschrift Wageningen Met lit. opg. - Met samenvatting in het Engels en Nederlands, s.n.], [S.l.

UNEP-IETC. (2004). *Waste management plnning: An environmentally sound approach for sustainable urban waste management*, *An introductory guide for decision-makers*: United Nations Publication.

UNEP. (2005). Selection, design and implementation of economic instruments in the solid waste management sector in Kenya: The case of plastic bags. Nairobi: United Nations Environment Programme.

USEPA. (1998). Landfill air emissions estimation model (Version 2.01), EPA-68-D1-0117, EPA 68-D3-0033, U.S Environmental Protection Agency.

USEPA. (2002). Solid Waste Management and Greenhouse Gases 2nd Ed.: EPA530-R-02-006. United States Environmental Protection Agency (U.S. EPA).

USEPA. (2005a). Direct Emissions from Landfilling Municipal Solid Waste. Climate leaders greenhouse gas inventory protocol core module guidance EPA-600/R-05/047, U.S Environmental Protection Agency.

USEPA. (2005b). First-Order Kinetic Gas Generation Model Parameters for Wet Landfills. EPA-600/R-05/072, U.S Environmental Protection Agency.

Valencia, R. V. (2008). Enhanced stabilization of municipal solid waste in bioreactor landfills, UNESCO-IHE Institute for Water Education, Technical University, Delft, The Netherlands.

Valencia, R., W. van der Zon, H. Woelders, H. J. Lubberding, and H. J. Gijzen. (2009). The effect of hydraulic conditions on waste stabilisation in bioreactor landfill simulators. *Bioresource Technology* 100 (5):1754-1761.

van Buuren, J.C.L. (2010). *Sanitation choice involving stakeholders: A participatory multi-criteria method for drainage and sanitation system selection in developing cities applied in Ho Chi Minh City, Vietnam*: Wageningen University.

van de Klundert, A., and J. Anschütz. (2001). *Integrated sustainable waste management - The Concept*. Edited by A. Scheinberg, *Tools for Decision-makers*. Gouda, The Netherlands.

Veeken, A. H. M. , and B. V. M. Hamelers. (2000). Effect of substrate-seeding mixing and leachate recirculation on solid state digestion of biowaste. *Water Science and Technology* 41 (3):255-262.

Veeken, A., S. Kalyuzhnyi, H. Scharff, and B. Hamelers. (2000). Effect of pH and VFA on hydrolysis of organic solid waste. *Journal of Environmental Engineering-Asce* 126 (12):1076-1081.

Vigneron, V., M. Ponthieu, G. Barina, J. M. Audic, C. Duquennoi, L. Mazéas, N. Bernet, and T. Bouchez. (2007). Nitrate and nitrite injection during municipal solid waste anaerobic biodegradation. *Waste Management* 27 (6):778-791.

Visvanthan, C., Nang Htay Yin, and Obuli P. Karthikeyan. (2010). Co-disposal of electronic waste with municipal solid waste in bioreactor landfills. *Waste Management* 30 (12):2608-2614.

Vymazal, J., and L. Kröpfelova. (2009). Removal of organics in constructed wetlands with horizontal sub-sruface flow: a review of the field experience. *Science of the Total Environment* 407:3911-3922.

Wang, ChihCheng, P. H. Lee, Kumar Mathava, YuTzu Huang, S. Sung, and JihGaw Lin. (2010). Simultaneous partial nitrification, anaerobic ammonium oxidation and denitrification (SNAD) in a full-scale landfill-leachate treatment plant. *Journal of Hazardous Materials* 175 (1/3):622-628.

Warith, M. A. (2002). Bioreactor landfills: Experimental and field results. *Waste Management* 22 (1):7-17.

Williams, P. T. . (2005). *Waste treatment and disposal*. 2nd ed. Great Britain: John Wiley & Sons.

Wiszniowski, J., D. Robert, J. Surmacz-Gorska, K. Miksch, and J. V. Weber. (2006). Landfill leachate treatment methods: A review. *Environmental Chemistry Letters* 4 (1):51-61.

Yalcuk, A., and A. Ugurlu. (2009). Comparison of horizontal and vertical constructed wetyland systems for landfill leachate treatment Bioresource Technology. 100 2521-2526.

Yuen, S. T. S., Q. J. Wang, J. R. Styles, and T. A. McMahon. (2001). Water balance comparison between a dry and a wet landfill -- a full-scale experiment. *Journal of Hydrology* 251 (1-2):29-48.

Zacharof, A. I., and A. P. Butler. (2004). Stochastic modelling of landfill processes incorporating waste heterogeneity and data uncertainty. *Waste Management* 24 (3):241-250.

Zamorano, M., J. Ignacio Perez Perez, I. Aguilar Paves, and A. Ramos Ridao. (2007). Study of the energy potential of the biogas produced by an urban waste landfill in Southern Spain. *Renewable and Sustainable Energy Reviews* 11 (5):909-922.

Zhao, X., R. Musleh, S. Maher, M. V. Khire, T. C. Voice, and S. A. Hashsham. (2008). Start-up performance of a full-scale bioreactor landfill cell under cold-climate conditions. *Waste Management* 28 (12):2623-2634.

Zhao, Xu, JiuHui Qu, HuiJuan Liu, ChunXia Wang, ShuHu Xiao, RuiPing Liu, PengJu Liu, HuaChun Lan, and ChengZhi Hu. (2010). Photoelectrochemical treatment of landfill leachate in a continuous flow reactor. *Bioresource Technology* 101 (3):865-869.

Zhao, Y. C., L. C. Wang, R. H. Hua, D. M. Xu, and G. W. Gu. (2002). A comparison of refuse attenuation in laboratory and field scale lysimeters. *Waste Management* 22 (1):29-35.

Zuilen, L. F. (2006). Planning of an integrated solid waste management system in Suriname: a case study in Greater Paramaribo with focus on households. PhD thesis, Applied Biological Science, Ghent University, Ghent.

About the Author

Fredrick Mathew Salukele was born on 11th December, 1974 in Arusha, Tanzania. In 2001 he got his Bachelor of Science degree in Environmental Engineering from UCLAS - University College of Lands and Architectural Studies (a constituent college of University of Dar es Salaam), Tanzania. He immediately joined UCLAS (currently Ardhi University) and became part of the academic staff in the department of Environmental Engineering until to date. In 2005 he was conferred a degree of Master of Science in Environmental Engineering from University of Dar es Salaam. In 2007, he joined for his PhD studies at sub department of Environmental Technology in Wageningen University, The Netherlands under the Partnership for Research on Viable Environmental Infrastructure Development in East Africa (PROVIDE) Project funded by INREF. He followed his PhD studies under SENSE graduate school.

His address in Tanzania:

Fredrick Salukele

Dept. of Environmental Engineering,
ARDHI UNIVERSITY,
P.O. Box 35176,
Dar es Salaam, Tanzania.
Email: fredrick.salukele@gmail.com; salukele@aru.ac.tz

Netherlands Research School for the
Socio-Economic and Natural Sciences of the Environment

C E R T I F I C A T E

The Netherlands Research School for the
Socio-Economic and Natural Sciences of the Environment
(SENSE), declares that

Fredrick Mathew Salukele

born on 11 December 1974 in Arusha, Tanzania

has successfully fulfilled all requirements of the
Educational Programme of SENSE.

Wageningen, 4 September 2013

the Chairman of the SENSE board

Prof. dr. Rik Leemans

the SENSE Director of Education

Dr. Ad van Dommelen

The SENSE Research School has been accredited by the Royal Netherlands Academy of Arts and Sciences (KNAW)

KONINKLIJKE NEDERLANDSE
AKADEMIE VAN WETENSCHAPPEN

The SENSE Research School declares that **Mr. Fredrick Salukele** has successfully fulfilled all requirements of the Educational PhD Programme of SENSE with a work load of 33 ECTS, including the following activities:

SENSE PhD Courses

- Environmental Research in Context
- Research Context Activity: Co-organizing of SENSE T-Bar meeting on Sustainable Anaerobic Technology; now and in the future. Wageningen, 8 June 2007
- Basic Statistics

Other PhD and Advanced MSc Courses

- Introductory course to the PROVIDE Project
- Urban Water and Waste: Technology and reuse
- Information Literacy

Management Skills Training

- Co-organizing the 3rd PROVIDE Conference *Strengthening Scientific Output - Socio-economic instruments and the MCDA method*, Arusha, Tanzania, 2008.

Oral Presentations

- *Solid waste diagnosis of rapidly growing cities the case of Mwanza city, Tanzania.* Strengthening Scientific Output - Socio-economic instruments and the MCDA method , PROVIDE, 6-8 October 2008, Arusha, Tanzania
- *Anaerobic treatment of solid waste and landfill leachate using Landfill Bioreactor technology.* Sanitation at System Level: Making sense of the results, 22-24 June 2009, Kampala , Uganda
- *Municipal Solid Waste Management Diagnosis of Rapidly Growing Cities: The Case Of Mwanza City, Tanzania.* 26th International Conference on Solid Waste Technology and Management, 27-30 March 2011, Philadelphia, U.S.A.
- *Testing and design of sustainable sanitation and solid waste technological options: Technological interventions of PROVIDE project*. INREF conference 2012 "Tackling the Development Challenges: The struggle for interdisciplinarity", 23-25 April 2012, Wageningen, The Netherlands

SENSE Coordinator PhD Education

Drs. Serge Stalpers