AN INVESTIGATION INTO THE EMISSIONS OF GREENHOUSE GASES ASSOCIATED WITH THE DISPOSAL OF SOLID WASTE IN THE ETHEKWINI MUNICIPALITY

by

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Thesis submitted in fulfilment of the academic requirements for the degree of

Doctor of Philosophy

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November 2013

Preface

The work described in this thesis was carried out through the School of Engineering, Programme of Civil Engineering, University of KwaZulu-Natal, Durban, from June 2007 to June 2013, under the supervision of professor Cristina Trois.

These studies represent original work by the author and have not otherwise been submitted in any form for any degree or diploma to any tertiary institution. Where use has been made of the work of others it is duly acknowledged in the text.

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Declaration 1 – Plagiarism

I, Elena Friedrich, declare that:

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Declaration 2 – Publications

All the papers listed in this section have been authored by myself, Elena Friedrich, with my supervisor Professor Cristina Trois being the second author. The work I did for these papers included: the conceptualisation of each paper, the planning of each paper, literature reviews, data collection, data analysis, calculations, modelling and the writing of all the papers. My supervisor, Professor Cristina Trois provided comments on the finalised manuscripts before submission. These comments were of editorial nature. For paper number three (3) Mr Chris Brouckaert provided help with the Monte Carlo simulations and one anonymous reviewer had inputs with regard to accounting of carbon stored in landfills. These contributions were also acknowledged at the end of this article and have been published.

Details of publications:

1. **Friedrich, E.** and Trois, C. (2010) Greenhouse Gas Accounting and Reporting for Waste Management – A South African Perspective, Waste Management, 30 (11), 2347-2353
2. **Friedrich, E.** and Trois, C. (2011) Quantification of Greenhouse Gas Emissions from Waste Management Processes for Municipalities – A Comparative Review Focusing on Africa, Waste Management, 31 (7), 1585-1596
3. **Friedrich, E**. and Trois, C. (2013) GHG Emission Factors Developed for the Collection, Transport and Landfilling of Municipal Waste in South African Municipalities, Waste Management, 33 (4), 1013-1026
4. **Friedrich, E.** and Trois, C. (2013) GHG Emission Factors Developed for the Recycling and Composting of Municipal Waste in South African Municipalities, Waste Management, accepted and in press
5. **Friedrich, E.** and Trois, C. (2013) Current and Future GHG Emissions from the Management of Municipal Solid Waste in the eThekwini Municipality – South Africa , to be submitted in November 2013.

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Acknowledgements

I would like to thank all the people who have helped and motivated me during my doctoral study.

I am greatly indebted to my family for their patience and encouragement during the years that this research took place. In particular, my husband Professor Holger Friedrich, was very supportive and on many occasions proof read my outputs and took care of the children so that I had the time and space to complete this work.

Many thanks go to all the people who assisted me with this research by giving some of their precious time, information and advice. This includes my supervisor Professor Cristina Trois as well as the staff from the City of Cape Town Waste Unit, The Glass Recycling Company, Plastics SA, Consol Glass, the Aluminium Federation of South Africa, Mpact Recycling, Mondi Paper and many others. Special thanks go to Mr John Parkin and Mr Logan Moodley from Durban Solid Waste, as well as to Mr Iain Kerr form the Paper Manufacturing Association of South Africa for providing data and always being available for an exchange of ideas.

I dedicate this work to my children Stefan and Lukas.

"*We do not inherit the earth from our ancestors; we borrow it from our children*." (Native American Proverb)

Abstract

The amount of greenhouse gases (GHG) emitted due to waste management in the cities of developing countries is predicted to rise considerably in the near future; however, these countries have a series of problems in accounting and reporting these gases. This study investigated GHG emissions from the municipal waste sector in South Africa. In particular, the eThekwini Municipality is researched in detail and current emissions as well as further projections have been calculated. This research has to be placed in the wider context where developing countries (including South Africa) do not have binding emission reduction targets, but many of them publish different greenhouse gas emissions data which have been accounted and reported in different ways. Results from the first stages of this research showed that for South Africa, inventories at national and municipal level are the most important tools in the process of accounting and reporting greenhouse gases from waste. However, discrepancies in the methodology used are a concern. This is a challenging issue for developing countries, especially African ones, since higher accuracy methods are more data intensive. Therefore, the development of local emission factors for the different waste management processes is important as it encourages a common, unified approach.

In the accounting of GHG from waste at municipal level, emission factors, based on a life cycle approach, are used with increased frequency. However, these factors have been calculated for many developed countries of the Northern Hemisphere and are generally lacking for developing countries. The second part of this research showed how such factors have been developed for waste processes used in this country. For the collection and transport of municipal waste in South Africa, the average diesel consumption is around 5 dm^3 (litres) per tonne of wet waste and the associated GHG emissions are about 15 kg CO_2 equivalents (CO_2 e). Depending on the type of landfill, the GHG emissions from the landfilling of waste have been calculated to range from -145 to 1 016 kg CO_2 e per tonne of wet waste, when taking into account carbon storage, and from 441 to 2 532 kg CO_2 e per tonne of wet waste, when carbon storage is left out. The highest emission factor per unit of wet waste is for landfill sites without landfill gas collection and these are the dominant waste disposal facilities in South Africa. The emission factors developed for the different recyclables in the country showed savings varying from -290 kg CO_2 e (glass) to – 19 111 kg CO_2 e (metals - Al) per tonne of recyclable. They also illustrated that there is variability, with energy intensive materials like metals having higher GHG savings in South Africa as compared to other countries. This study also showed that

composting of garden waste is a net GHG emitter, releasing 172 and 186 kg CO_2 e per tonne of wet garden waste for aerated dome composting and turned windrow composting, respectively.

By using the emission factors developed, the GHG emissions from municipal waste in the eThekwini Municipality were calculated and showed that for the year 2012 net savings of -161 780 tonnes CO_2 e were achieved. This is mainly due to the landfill gas to electricity clean development mechanism (CDM) projects and due to recycling in the municipality. In the absence of landfill gas (LFG) collection and utilisation systems, which is typical for the majority of South African landfills, important GHG emission from the anaerobic degradation of waste are recorded. In the near future (year 2014) the closure of one of the three local landfill sites and the re-directioning of the majority of waste to another landfill sites which does not have LFG collection and utilisation, will cause an increase of GHG emissions to 294 670 tonnes CO_2 e. An increase in recycling and the introduction of anaerobic digestion and composting has the potential to reduce these emissions as shown for the year 2020. However, only the introduction of a LFG to electricity system will result in the highest possible overall GHG savings from waste management in the municipality. In the absence of the Clean Development Mechanism and the associated financial arrangements, these systems have to be financed locally and might present a financial challenge to the municipality. Therefore, the second intervention which will make a difference by lowering GHG emissions from waste management would be to increase recycling in general and in particular the recycling of paper and metals. Since there is no direct competition for carbon, in addition to recycling, anaerobic digestion can be introduced and this combination will achieve increased savings in the future. If anaerobic digestion is not possible, composting in addition to recycling will also lead to savings, albeit not as high as with anaerobic digestion.

The results presented in this study show that life cycle based GHG emission factors for waste and their use can support a unified approach to accounting of GHG and better decision-making for municipalities in the local context. They can give valuable input for the planning and development of future waste management strategies and they can help optimise current municipal solid waste management.

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Chapter 1 Overall Introduction

1.1 Background Information

Global climate change is the most important environmental problem currently facing mankind and it is caused by the anthropogenic release of greenhouse gases (GHG) into the Earth's atmosphere (IPCC, 2007). Global GHG emissions due to human activities have increased exponentially since pre-industrial times. The most important anthropocentric GHG are carbon dioxide (CO_2) and methane (CH_4), followed by nitrous oxide (N_2O). The tropospheric concentration of all three gases showed marked increases in the last decades, with carbon dioxide increasing from a concentration of 280 to 400 parts per million (US NOAA, 2013) and methane increasing from a concentration of 700 to 1870 parts per billion (CDIAC, 2013). Similarly nitrous oxide increased from 270 to 324 parts per billion (CDIAC, 2013). Compared to carbon dioxide, methane and nitrous oxide are more potent in terms of global warming potential, causing gram for gram in the time frame of 100 years an effect which is 25 and 298 times higher respectively, than that of carbon dioxide (IPCC, 2007). If current emission trends for these gases are continuing it is expected that the earth's surface temperature will increase substantially in the future and will lead to important changes in the global climate system (IPCC, 2007), with far reaching consequences for humanity and the planet. Therefore, there is a need to quantify, control and reduce the emission of GHG from each sector, including from the management of waste.

South Africa is the 12th largest emitter of GHG globally and an estimated 387 million metric tonnes CO_2 equivalents (CO_2 e) were released in 2004, representing 1.6 % of global emissions (RSA National Treasury, 2010). About 2 % of the South African GHG emissions are due to the waste sector (Greenhouse Gas Inventory, 2009). On a global scale Bogner et al. (2008) estimated that the GHG emissions from post-consumer waste and wastewater contribute to about 3% of total anthropocentric GHG emissions and totalled about 1.4 Gt CO_2 e for the year 2004-2005. The majority of these emissions (90%) consist of methane emitted from landfill sites (Bogner et al., 2007). The CH_4 emissions from the waste sector constitute about 18% of the global anthropocentric CH_4 emissions (Bogner et al., 2007). Nitrous oxide, carbon dioxide of fossil origin and halogen-containing gases from waste management also cause global warming but they are considered minor contributors (Bogner et al., 2007). In this context it is important to differentiate between biogenic and fossil CO_2 from waste management. CO_2 from biogenic sources (e.g. food and garden waste) is considered neutral to climate change because it is part of the short carbon cycle. However, fossil CO_2 is not neutral and contributes to global warming.

1.2 Motivation for the Research

In spite of the relatively minor contribution that the waste sector makes to national and global GHG emissions, it is important to investigate this contribution because "*the waste sector is in the unique position to move from being a minor source of global emissions to becoming a major saver of emissions*" (UNEP, 2010). Scheutz et al. (2009) summarise the most important GHG sources and sinks associated with the waste management industry for each of the three life cycle stages possible, namely upstream indirect emissions (e.g. CO_2, CH_4 and N_2O from the collection and transport of waste), operating direct emissions (e.g. CH_4 from landfilling) and downstream indirect emissions (e.g. avoided GHG emissions due to substitution of raw materials due to recycling). In particular, downstream indirect emissions are associated with materials and energy savings and have the potential for significant GHG emission savings due to waste management. However, the same authors also underline that there is much uncertainty in quantification for all these life cycle stages (Scheutz et al., 2009). In South Africa this uncertainty is amplified due to the paucity of data collected and available for all waste management activities. Godfrey (2008) showed that only 68.9% of the South African municipalities surveyed collected some data on waste management, and the type and quality of these data varied considerably.

The quantification of greenhouse gases has to be seen in a wider context in which a waste management system operates within and interacts with other systems of a country or region. The quality of carbon balances and life cycle assessment studies for waste (and the resultant emission factors) depend on the availability and quality of data regarding waste management, but also on data on the energy system of that country or region. South Africa, as do many other developing countries (e.g. China and India), relies heavily on coal for energy generation. Therefore, any waste management process which will save (e.g. source reduction and recycling) or replace (e.g. landfill gas to electricity) this fossil fuel intensive energy will have positive effects in terms of overall GHG emission reductions. Some of these aspects are fundamental to the Clean Development Mechanism (CDM) projects for Non-Annex 1 countries (i.e. landfill gas to electricity). However, other aspects (i.e. recycling and transport of waste as part of an integrated waste management system) have not been well investigated locally in terms of associated GHG emissions. In addition, because of different capacities of local South African authorities, these investigations are not always possible, especially in the smaller municipalities in rural areas, which lack resources and know-how to perform even the basic waste management functions, let alone GHG accounting. Even in the large municipalities which do record waste data, the process of quantifying and accounting GHG from waste management is generally targeting landfill sites emissions (due to the CDM requirements) and not the entire waste management system. By ignoring the emissions from other processes in the waste management system (e.g. transportation, recycling, waste minimisation) and by focussing mainly on processes which bring revenue (i.e. landfills with CDM projects) the overall optimisation of these systems may be jeopardised and important avenues for GHG reductions from waste ignored. Therefore, the overall aim of this research was to develop a South African approach for GHG accounting from waste management in order to perform a better quantification of these emissions for all processes and a uniform application within the waste sector.

1.3 Research Question, Aims and Objectives

This study aimed to take a holistic, integrative approach at characterising the greenhouse gas emissions due to the disposal of solid waste in South Africa and to provide guidelines on local waste management methods for the reduction of greenhouse gas emissions. Therefore, the research question has been defined as:

What are the best local waste disposal practices/strategies that will ensure an effective reduction of greenhouse gas emissions from the management of municipal solid waste?

The ancillary question was:

Can the municipal waste sector become a saver of greenhouse gas emissions?

The objectives of the study were as follows:

1. Review existing information on the quantification of GHG from the management of municipal solid waste, emphasising the situation in developing countries and in particular in Africa and South Africa.

2. Review current GHG accounting and reporting from the waste sector at local and national level.
3. Conceptualisation of a system approach to quantify greenhouse gases based on local emission factors.
4. Develop the South African GHG emission factors for the different waste management processes used in the country.
5. Using the above factors to calculate current and future GHG emissions from waste management for the eThekwini Municipality and to investigate GHG reduction opportunities.

The main aim of the study was to provide some guidelines on the best local waste management methods and the supporting operational systems with regard to reduction of greenhouse gas emissions. This study intended to fill the important knowledge gap on emissions due to management of municipal solid waste and to provide a basis for the design of a protocol of best practice for municipalities towards the reduction of carbon emissions.

1.4 Technical and Scientific Contributions

The most important contribution emerging from this research is the development of GHG emission factors for the processes used for solid waste management in South Africa. These are presented in Chapter 4 of the dissertation. These factors enable a unified approach to accounting of GHG from waste and can lead to better decision-making for municipalities in the local context as has been shown in Chapter 5. They can give valuable input for the planning and development of future waste management strategies and they can help optimise current municipal solid waste management. Furthermore, during the final stages of this research the author found out that their application has been extended to private South African companies. Just after the publication of the GHG emission factors for landfilling (Paper 3 in the publication list and the first part of Chapter 4) they were used by a Cape Town consultancy to calculate scope 1, 2 and 3 emissions from waste management for a local company.

1.5 Structure of the Thesis

This thesis is organised into six chapters, of which five contain reproductions of papers which are presented in their original form. An overall introduction is given in this chapter and a summary of the findings and overall conclusions are provided in Chapter 6. The other chapters of the thesis are presented in the following paragraphs.

Chapter 2 contains the literature review. This review summarizes and compares GHG emissions from individual waste management processes which make up a municipal waste management system, with an emphasis on developing countries and, in particular, Africa. It should be seen as a first step towards developing more holistic GHG accounting for municipalities.

Chapter 3 investigates how greenhouse gases are accounted and reported in the waste sector in South Africa. Developing countries (including South Africa) do not have binding GHG emission reduction targets, but many of them publish different greenhouse gas emissions data which have been accounted and reported in different ways. For local waste management these accounting and reporting tools are investigated in this chapter for South Africa (national level) and for the eThekwini Municipality (municipal level).

Chapter 4 presents the GHG emission factors developed for waste management in this country. The first part of this chapter reports the GHG emission factors for collection, transport and landfilling of municipal waste and the second part reports these factors for recycling and composting. Glass, metals (Al and Fe), plastics and paper were chosen, as these are the main recyclables.

Chapter 5 presents current emissions and future projections of GHGs from municipal solid waste management for one metropolitan area in the country, namely, the eThekwini Municipality. In addition it investigates consequences of planned changes in the current municipal waste management system and possible future scenarios which can lead to lower GHG emissions feasible in the local context.

Chapter 6 summarises the key findings from the study, presents final conclusions and provides recommendations for further research.

It should be noted that there is some repetition in the chapters which have been published as separate, standalone papers. In particular, information on the waste management in the eThekwini Municipality had to be repeated in different forms in three of these papers.

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Chapter 2 Quantification of Greenhouse Gas Emissions from Waste Management Processes for Municipalities – A Comparative Review Focusing on Africa

2.1 Abstract

The amount of greenhouse gases (GHG) emitted due to waste management in the cities of developing countries is predicted to rise considerably in the near future; however, these countries have a series of problems in accounting and reporting these gases. Some of these problems are related to the status quo of waste management in the developing world and some to the lack of a coherent framework for accounting and reporting of greenhouse gases from waste at municipal level. This review summarizes and compares GHG emissions from individual waste management processes which make up a municipal waste management system, with an emphasis on developing countries and, in particular, Africa. It should be seen as a first step towards developing a more holistic GHG accounting model for municipalities. The comparison between these emissions from developed and developing countries at process level, reveals that there is agreement on the magnitude of the emissions expected from each process (generation of waste, collection and transport, disposal and recycling). The highest GHG savings are achieved through recycling, and these savings would be even higher in developing countries which rely on coal for energy production (e.g. South Africa, India and China) and where non-motorized collection and transport is used. The highest emissions are due to the methane released by dumpsites and landfills, and these emissions are predicted to increase significantly, unless more of the methane is captured and either flared or used for energy generation. The Clean Development Mechanism (CDM) projects implemented in the developing world have made some progress in this field; however, African countries lag behind.

2.2 Introduction

With global warming becoming an important environmental issue, many studies have investigated the topic of greenhouse gas emissions (GHG) from waste activities (Kennedy et al., 2009; Gentil et al., 2009; Friedrich and Trois, 2010). It is estimated that the post-consumer waste sector contributes about 3 to 4% to the total global anthropogenic GHG emissions and for 2004-2005 this contribution amounted to $49*10^9$ tonnes CO_2 e per year (Bogner et al., 2008). Although this contribution is considered relatively small, the carbon reduction opportunities for the sector are still not fully explored (ISWA, 2009), in particular in developing countries. In the year 2000, developing countries were responsible for about 29% of these emissions and this share is predicted to increase to 64% in 2030 and 76% in 2050 with landfills being the major contributor to this increase (Monni et al., 2006). A series of initiatives were highly successful and showed that large reductions in emissions are possible. For example, the contribution of the European municipal waste sector decreased from $69*10^6$ tonnes CO_2 e in 1990 to $32*10^6$ tonnes CO_2 e in 2007 and further reductions are projected (ISWA, 2009). The situation in developing countries is different and has to be changed if overall emissions are to be stabilized. Under the Kyoto Protocol, developing countries do not have any mandatory obligations to reduce GHG emissions, however, there are many voluntary and carbon market driven initiatives in this direction. In this context "*accurate measurements and quantification of greenhouse gas emissions is vital in order to set and monitor realistic reduction targets at all levels*" (ISWA, 2009).

In general, the majority of studies investigating the emissions of greenhouse gases from waste focused on individual waste management stages (especially waste disposal through landfilling) and other processes, in particular waste minimization and transport of waste, were not always included. Furthermore, developing countries, which due to their population sizes are important generators of municipal waste, have been less researched than their developed counterparts. As a result a more systemic and holistic approach is needed for developing countries. In this context the entire waste management system needs to be considered to properly evaluate the best strategies to reduce greenhouse gases and to assess how different waste management processes can be combined and optimized for this purpose. This is of particular importance at local level, since local authorities are in charge of managing waste on a daily basis and they are the primary agents when planning and enforcing changes. Yet for local authorities there are no clear rules and/or guidelines on how to account and report greenhouse gases from waste. Five different methodological approaches have been presented in the literature (Kennedy et al., 2009) and have been used by cities. They differ mainly by the processes of the waste management system which they include and by using different time frames for the calculation of emissions. Therefore, published GHG emissions figures from waste for municipalities cannot be compared between the different studies (sometimes for the same municipality), which make approaches towards improvement difficult to develop and assess.

The amounts of waste generated, the composition of the waste (in particular the carbon content) as well as the technologies used for handling and disposing this waste will determine the final amount of greenhouse gases emitted from a waste management system. A comparative analysis of the published literature showed that all important factors vary between developing and developed countries and they have been differently incorporated in the different accounting techniques for the waste sector (Friedrich and Trois, 2010, Couth and Trois, 2010 and 2011). The aim of this paper is to summarize and compare the existing literature on the quantification of greenhouse gases from waste at municipal level in developed and developing countries with a particular focus on the African continent and South Africa. This should be the first step in the development of more holistic quantification models and overall strategies to reduce these emissions. It also aims to identify gaps and problematic areas for quantifying GHG emissions in developing countries and in particular in Africa. As such it investigates individual processes in the waste management cycle, starting with the generation and composition of waste, followed by collection and transport, disposal processes and recovery and recycling.

2.3 Overview of GHG quantification models for municipal waste and their relationship to the waste management processes

In an overview article, Gentil et al. (2009) described the four main types of GHG accounting methodologies in waste management as national accounting (with reference to the IPCC method), corporate level accounting (including local government, i.e. municipalities), life cycle assessment and carbon trading methodologies. At municipal level all of these four types of accounting methodologies can and have been employed in different investigations, even though the IPCC model has been designed for national use. The GHG accounting results differ greatly between these methodologies based on what was included and what was left out. To make the process of accounting and reporting more transparent, Gentil et al. (2009) propose the upstream-operating-downstream conceptual framework. In this context it is important to acknowledge that "*the choice of GHG accounting mechanism depends on the scope of the reporting, but all rely on the same basic operational data generated by the individual waste management technologies*" (Gentil et al., 2009). As a result there is a need to investigate the relationship between the accounting tools used for GHG emissions from municipalities and the actual processes/technologies which give rise to these emissions.

In general, the relationships between the quantification approach (or technique) used and the waste management processes, which make up a particular waste management system can be schematically represented as in Figure 2.1. As presented in this figure there are two other important factors which shape the quantification process, namely the motives and drivers for reporting and the availability of data on the processes included in the waste management system. These factors are different in developed and developing countries, with developed countries' mandatory obligation to report greenhouse gases and therefore, the need to collect, model, calculate and/or validate data on waste management. Developing countries do not have such an obligation and their reporting process is voluntary and the availability of data is much reduced.

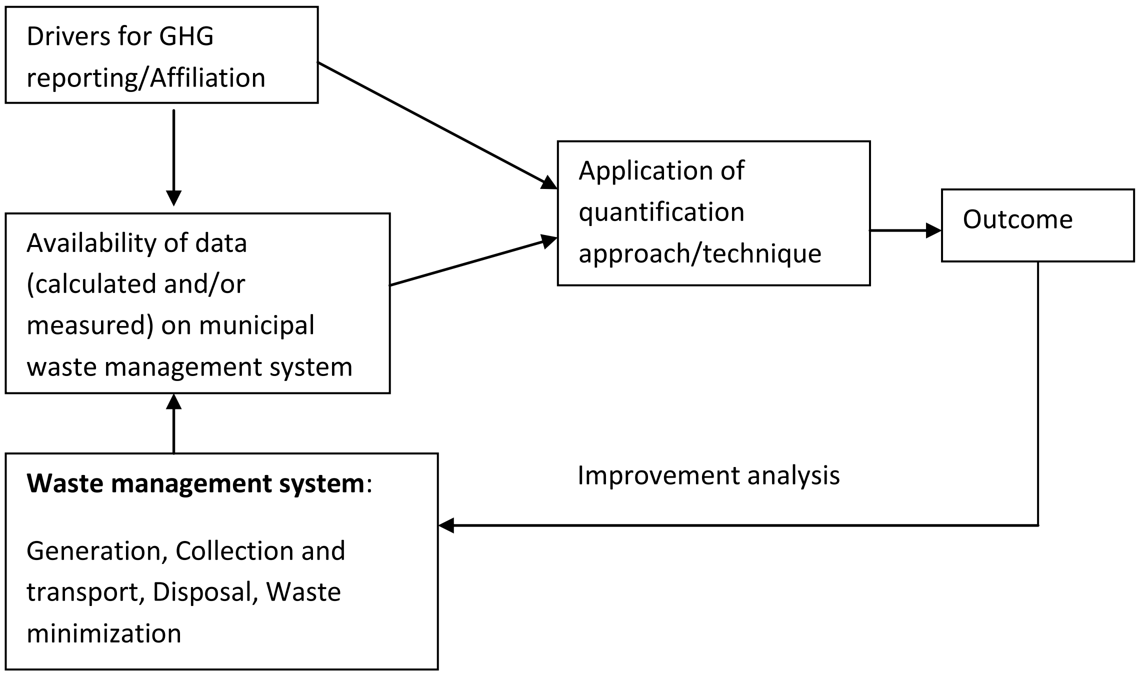

Figure 2-1: General framework for greenhouse gas quantification for municipal waste

With regard to the application of the quantification techniques in municipalities in developing countries there are important factors influencing the outcome. These factors are related to the inherent problems developing countries are facing with regard to waste management (i.e. lack of resources, expertise, information etc.) and they have been mentioned extensively in the literature (e.g. Henry et al., 2006; Matete and Trois, 2008; Manga et al., 2008). These factors are also affecting the availability of data and they lead to inefficient outcomes in terms of evaluation of the GHG from waste. This situation has somewhat changed with the implementation of the clean development mechanism (CDM) projects and the more rigorous calculation and validation methods they require (Couth and Trois, 2011 and Friedrich and Trois, 2010). However, by focusing only on one component/process of the waste management system (i.e. the component which earns carbon credits) better opportunities in terms of the overall outcome might be neglected. Therefore, individual processes within the waste management system have to be individually researched, but they must be seen also as parts of the system. The following sections present a more detailed review on individual processes and their determining factors in terms of GHG emissions. In this paper whenever figures are presented they refer to a mass unit of wet waste (per tonne), unless otherwise specified.

2.4 The generation and composition of waste and the potential for GHG emissions

The first component in any waste management system is the amount of waste generated and the nature of that waste. This is also important in terms of the quantities of greenhouse gases to be generated from that waste. Waste generation has been correlated in the literature to population size, wealth and urbanization (Bogner et al., 2008; Cointreau, 2006). The rate of increase in waste generation shows that both developing and developed countries increased their waste generation per capita, however, some countries have higher generation trends (OECD Factbook 2009). European countries seem to have stabilized their waste generation rates and are moving towards de-coupling waste generation from economic growth (Mazzanti and Zoboli, 2008), whereas other countries, including developing countries (but also some developed countries) continue to show marked growth in the amounts of waste produced. One such country, in particular, is China (Zhang et al., 2010). In addition, a review of the absolute values showed that, due to their large populations, even if waste generation rates per capita are low, developing countries produce large amounts of waste (OECD Factbook 2009). These amounts are expected to rise with increased urbanization and consumerism, even in cities in the developing countries that have high poverty rates. Barton et al. (2008) calculate, based on data from Shimura et al. (2001), that about $226*10^6$ tones per year of waste will be produced by the one billion people living in the slums of the developing world. They assume an average generation rate of 0.6 kg per capita per day (or 219 kg per capita per year).

Troschinetz and Mihelcic (2009) presented a comparative analysis of the waste generation rates for 23 developing countries and the OECD (1.43 kg/capita/day), European Union (1.51

kg/capita/day) and United States (2.08 kg/capita/day). Table 2.1 presents more detailed waste generation rates currently available for developing countries.

Table 2-1: Recent published waste generation data for developing countries.

City/Country	Amount generated (kg/capita/day)	Biodegradable organic fraction (%)	Reference
DEVELOPING COUNTRIES			
Allahabad - India	0.39	45,3	Sharholly et al., 2007
Indian cities - Review	0.2-0.5	40-60	Sharholly et al., 2008
Puducherry - India	0.59	65	Pattnaick and Reddy, 2010
*IPCC rate for India**	*0.46*	-	*IPCC, 2006*
Chittagong - Bangladesh	0.25	66	Sujauddin et al., 2008
Beijing - China	0.23	69.3	Qu et al., 2009
Pudong (Shanghai) - China	1,11	59	Minghua et al., 2009
Chongquing - China	1.08	59	Hui et al., 2006
*IPCC rate for China**	*0.75*		*IPCC, 2006*
Kuala Lumpur - Malaysia	1.5	68.6	Saeed et al., 2009
Rasht - Iran	-	80,2	Moghadam et al., 2009
Nablus district - Palestine	0.82	65.1	Al-Khatib et al., 2010
Teheran - Iran	0.88	42.6	Damghani et al., 2008
Cape Haitian - Haiti	0.21	65.5	Philippe and Culot, 2009
Lahore - Pakistan	0.84	-	Batool and Ch, 2009
Kathmandu - Nepal	0.3 (calculated)	57.8	Alam et al., 2008
Cambodia	0.34	66	Parizeau et al., 2006
Chihuahua - Mexico	0.59	45	Gomez et al., 2009
Zarqa City - Jordan	0.44	56	Mrayyan and Hamdi, 2006
Southern Sri Lanka	0.27	66	Vidanaarachchi et al., 2006
Average developing countries	***0.58***	***59.4***	
AFRICA			
Makurdi - Nigeria	0.54	36-57	Sha'Ato et al., 2007
Abuja - Nigeria	0.55-0.58	52-65.3	Imam et al., 2008
Ibadan - Nigeria	0.2-0.33	-	Ayininuola and Muibi, 2008
Freetown – Sierra Leone	0.4-0.6	80	Ndomahina (2009) in Sankou et al., 2009
Accra - Ghana	0.4	60	Fobil et al., 2008
Dar es Salaam - Tanzania	0.4	-	Kaseva and Mbuligwe, 2005
Botswana	0.33	68	Bolaane and Ali, 2004
South Africa developed areas less developed areas	 0.8 0.3	-	Karani and Jewasikiewitz, 2006
*IPCC rate for Africa (based on a study from Sudan)**	*0.79*	-	*IPCC, 2006*
Average African countries	***0.44***	***59.8***	

** excluded from the calculations for the average*

From the analysis undertaken by Troschinetz and Mihelcic (2009) it is evident that the majority of developing countries included have much lower generation rates, but there are also exceptions (e.g. Maldives (2.48 kg/capita/day), Thailand (1.44 kg/capita/day in urban areas like Bangkok) and Mauritius (1.30 kg/capita/day)). These exceptions are due to specific circumstances, for example the Maldives and Mauritius are island states with a large tourist industry, while waste generation in Thailand is concentrated in large cities like Bangkok.

From Table 2.1 it can be seen that the waste generation rates per capita reported for African cities and countries are some of the lowest. This is also confirmed by Couth and Trois (2010) which took into account other published sources for generation rates. This has particular implications for GHG calculations, not only at municipal level but also at national level. In the calculations used to produce national inventories, regional waste generation rates as published by the IPCC (IPCC, 2006) are used by countries which do not have waste generation data. Most of the African countries are in this situation; although some studies which incorporated waste generation data have been published in recent years (see Table 2.1 – African countries). All reported rates from the literature on African case studies, as summarized in Table 2.1, are lower than the generation rate recommended by the IPCC (2006) methodology to be used for GHG national inventories for the African continent. It has to be underlined that municipal areas account for most of the greenhouse gases from waste of African countries and rural areas have a very low contribution (Couth and Trois, 2010). Most of the studies for the African countries included in Table 2.1 have been done for urban areas; therefore, the overall national generation rate for these countries could be even lower. In the view of the emerging literature as presented in Table 2.1, the IPCC 2006 generation rate for African countries calls for revision.

Of particular interest with regard to the potential for GHG generation has been the composition of the waste and in particular the biodegradable organic fraction which will ultimately give rise to greenhouse gases. In this context it is very important to distinguish biogenic carbon, which is not included in GHG inventories because it is seen as part of the natural carbon cycle (IPCC, 2006). Table 2.1 also summarizes the most recently published data on the biodegradable organic fraction in developing countries and in general this fraction is much higher in these countries as compared to their developed counterparts. More details are presented by Troschinetz and Mihelcic (2009). There are also other differenced in terms of waste composition, with developing countries having on average half as much paper and cardboard, as well as glass and plastic (Troschinetz and Mihelcic, 2009). The waste compositions from African cities and the developing world in general, tend to show high fractions in terms of organic, biodegradable materials (see Table 2.1 and also Couth and Trois (2010)).

As cities situated in the developing world become more affluent the composition of waste is expected to change. It has been observed that with an increase in the living standards the composition of waste also changes, with the biodegradable fraction resulting from unprocessed foods decreasing and an increase in paper, plastic, glass, textile and rubber (Cointreau, 2006; Moghadam et al., 2009 and Troschinetz and Mihelcic, 2009). The consequences of this change in terms of GHG emissions depend on the disposal methods used for the waste. Barton et al. (2008) showed that if the waste composition changed towards a more developed country composition, the amount of greenhouse gases increased in both open dump and landfill without gas collection scenarios (mainly due to the increase in the paper and textile fraction). These are the most frequently used disposal methods in the developing world, including Africa.

Troschinetz and Mihelcic (2009) showed that developing countries have a higher variance in the material fraction composition of all waste categories, but in particular for the organic fraction (due to seasonal factors, affluence, domestic fuel supply, geography, etc.). This underlines the dynamic complexities in modeling waste generation in developing countries and the need for regular studies with regard to waste composition and generation. However, due to lack of resources and management capabilities, these studies are less frequent in developing countries and, in particular, in Africa. This has particular implications for these countries and, in general, the accuracy of GHG calculations for these countries is lower.

2.5 Collection and transport of waste and the emission of greenhouse gases

Greenhouse gases are emitted in the collection and transport of waste from the combustion of fuel and mainly carbon dioxide, but also small amounts of other GHG (i.e. nitrous oxide and methane), are generated. Although these emissions are seldom included in GHG calculations for waste systems, it is necessary to acknowledge their contributions. In some waste management systems in developed countries (Salhofer et al., 2007) and some developing countries (Chen and Lin, 2008) they proved to be significant.

2.5.1 Factors important for GHG emissions in the waste collection process

There have been marked differences between the collection of waste in developed and developing countries, which in turn reflected on the GHG emissions from these processes. One of these differences is related to the collection rates of municipal waste. Collection rates have been much lower in developing countries as compared to their developed counterparts. For example OECD countries report collection rates varying between 90 and 100 % in their member countries (OECD, 2009), whereby developing countries have much lower rates as some of the examples presented in Table 2.2 are showing. Collection rates refer to the generated waste.

Table 2-2: Collection rates in selected developing countries/cities with emphasis on Africa

Country/Locality	Collection rate (%)	Reference
South Africa – general urban kerbside	50 80	DEAT, 2007 Karani and Jewasikiewitz, 2007
Abidjan (Côte d'Ivoire)	30-40	Parott et al. (2009)
Dakar (Senegal)	30-40	Parott et al. (2009)
Dar es Salaam (Tanzania)	48	Parott et al. (2009)
Lomé (Togo)	42.1	Parott et al. (2009)
Ndjamena (Chad)	15-20	Parott et al. (2009)
Nairobi (Kenya)	30-45	Parott et al. (2009)
Nouakchott (Mauritania)	20-30	Parott et al. (2009)
Yaoundé (Cameroon)	43	Parott et al. (2009)
China - general	79	Suocheng, 2001
Indian cities	About 70	Sharholy, 2008 and Pattnaik, 2010
Lahore (Pakistan)	60	Batool and Ch, 2009

Collection rates reported for African cities, except for South Africa, are much lower than those reported for other developing countries (see Table 2.2). These rates varied between 15 to 48 % for eight sub-Saharan cities as reported by Parrot et al. (2009), with Ndjamena (Chad) displaying the lowest collection rates of 15-20% and Dar es Salaam (Tanzania) the highest (around 48%). In the developing world there have been significant differences between waste collection in rural and urban areas, with the latter generally having higher collection rates. In addition, collection rates in African countries seem to have been more variable and fluctuated in time, not only geographically, improving dramatically like in the case of Accra (Ghana) where collection rates increased from 51 % in 1998 to 91% in 2000, (Fobil et al., 2008), but also deteriorating like in the recent case of Zimbabwe where economic hardship contributed to inadequate waste collection especially in low-income and informal areas (Nyathi, 2008).

In terms of GHG emissions, lower collection rates translated into lower emissions, since less transport was required and the degradation of un-collected municipal waste was assumed to be aerobic with no methane generation. Shimura et al. (2001) showed that the uncollected waste in cities in developing countries is either self-disposed (proper and improper), illegally dumped or recycled. For example, for Dar es Salaam (Tanzania) from the 1 772 tonnes per day waste generated, 654 tonnes per day (36,9%) were self-disposed, 847 tonnes per day (47,8%) were illegally dumped (of that 8.6% was dumped after it was collected), 130 tonnes per day (7,3%) were recycled, and only 143 tonnes per day (8.1%) were collected and disposed in a landfill site. Although the GHG emissions from this uncollected waste are expected to be lower, the other environmental impacts of municipal waste and its degradation products in a city environment (e.g. odours, ground water pollution, infestations, aesthetics, etc.) are relevant and have been extensively presented in the literature (e.g. Qasim and Chiang, 1994; Williams, 1998).

Developing countries are also employing different collection methods which might not be technologically advanced, but which in terms of GHG emissions have some advantages. For example, in many African cities manpower has been used in the collection of waste (e.g. for push carts, wheelbarrows, pedal tricycles, animal drawn carts etc.) (Imam et al., 2008) which avoids the use of fossil fuels and the resultant emissions. Similar use of manpower was also reported in many other developing cities (Rouse and Ali, 2002). Although low in technology, these positive aspects in terms of GHG emissions in the collection process should be encouraged and made more efficient in existing waste management systems, since they have not only social benefits (job creation) but also environmental ones. For example, it is estimated that around 5% of the jobs of urban poor in low-income countries are due to waste collection and transport (Rouse and Ali, 2002).

2.5.2 Factors important for GHG emissions in the transport process

In the international literature, drawing mainly from case studies in the first world, the GHG emissions associated with the transport of waste have been emanating from life cycle assessment type of studies. These studies showed that the most important factors in the transport and collection of waste with regard to GHG emissions are:

1. distances involved and the mode of transportation - with road transport having higher emissions per tonne of waste as compared to rail and barge transport (Salhofer et al., 2007 and Eisted et al., 2009),
2. population density of the area from where the waste was collected and transported – with densely populated areas being the most efficient ones (Larsen et al., 2009a), and
3. type of waste transported – with low density waste (e.g. expanded polystyrene) causing significant emissions (Salhofer et al., 2007).

For developing countries an additional factor which needs to be considered is the status quo of the vehicle fleet, which includes the age of the vehicles and their maintenance. Older vehicles and poor maintenance are associated with higher GHG emissions.

Transportation modeling studies showed that for loaded waste trucks the shortest route is not always the most fuel (and GHG emission) efficient one (Tavares et al., 2009) and road inclination and vehicle weight also played a role. There have been several recent studies investigating waste transport vehicles routing and the savings that can be achieved (Nguyen and Wilson, 2010; Arribas et al., 2010; Tavares et al., 2009, etc.). In general they showed that improvements can be achieved in terms of fuel efficiency (and associated emissions) by choosing the best routes and by changing the logistics and the organization (McLeod and Cherrett, 2008) of the waste collection and transport process. Nguyen and Wilson (2010) have calculated that for the trucks they had investigated in detail, more than 60% of the daily total fuel was consumed for the collection of waste and the transport accounts for the remaining fuel used (and associated emissions).

There have been very few studies investigating the GHG emissions from transporting waste in developing countries and the factors which are important for the developed world are only partially applicable in the developing countries. For example, population densities are seen as a factor decreasing transport emissions in first world countries (Larsen et al., 2009a and Nguyen and Wilson, 2010), however, in developing countries high population densities have been associated with un-planned informal settlements and an array of problems associated with the collection and transport of waste, e.g. the lack of access for vehicles (Fobil et al., 2008). With regard to the density of the waste, it is considered that waste densities in developing countries are higher than waste densities in developed countries and, therefore, sophisticated compactor trucks for collection and transport are not essential (Barton et al., 2008 and Imam et al., 2008) resulting in lower emissions. Low density waste fractions which have been associated with higher transport emissions make up only a small percentage of the waste in developing countries. For example, the average plastic contribution to the total waste stream for seven African studies is only 9.2% (Couth and Trois, 2010).

2.5.3 Quantification of GHG from collection and transport

The investigations of GHG emission from waste revealed that there is considerable variation in the amount of fuel and the resultant emissions per tonne of waste collected and transported (assumed to be wet waste). Larsen et al. (2009a) showed that for two municipalities in Denmark fuel consumption varied between 1.4 and 10.1 L diesel per tonne of waste collected and transported by road. Taking into account only GHG emissions from the collection and transport of waste a similar variability is reported in the literature. For developed countries a range of 5 to 50 kg of CO_2 e per tonne of wet waste was reported (Eisted et al., 2009), with European values at about 7.2 kg of CO_2 e per tonne of waste transported (Smith et al., 2001). For developing countries the GHG emissions calculated per tonne of waste were towards the lower ranges, with Chen and Lin (2008) reporting 16.38 kg of CO_2 e (or 4.47 kg C equivalents) per tonne of waste collected and transported in Taipei City (Taiwan), and Friedrich and Trois (2008) reporting 15.53 kg of CO_2 e per tonne of waste collected and transported in Durban (South Africa). These figures refer to wet waste. The waste collection and transportation processes for the two studies in the developing world were similar to those in the developed world (high collection rates, mechanized collection and transport and efficient local authorities) and might not be a true reflection on the majority of cities in the developing world and especially in Africa, where lower emissions (per tonne of waste) are expected due to the waste collection and transport process. On the other side, for the mechanized collection that exists in African countries, the age of the collection vehicles is much higher and there is lack of maintenance of the vehicle fleet which in turn might lead to higher emissions. For example, in Ibadan (Nigeria) the local authority had 45 collection vehicles servicing about 1.8 million people and 95% of these vehicles have been out of order, due to inadequate maintenance (Ayininuola et al., 2008). Henry et al. (2006) showed that in Kenya more than a third of the collection vehicles used by the largest five municipalities were out of service during the study year and most of the trucks were older than 10 years.

2.6 GHG emissions from waste disposal processes/technologies

The disposal of municipal waste in developing countries and in some of the developed countries is heavy reliant on landfills. It is considered that *the most common method of waste disposal in the developing countries is some form of landfilling* (UNEP, 2004) and this includes open uncontrolled dumps, as well as their more controlled and/or engineered counterparts. Other waste disposal technologies used in the developing world (i.e. incineration, composting, recycling and anaerobic digestion) has been also used and they also do produce GHG emissions. However, the amounts of greenhouse gases emitted due to these disposal technologies have been much lower and some of these processes could have important potential savings in terms of these emissions.

2.6.1 GHG from the decomposition of waste in landfills

The majority of studies investigating GHG emissions from waste management systems focused on landfills as the major contributing component, due to their methane emissions. Bogner et al.(2008) estimate that about $1.4 * 10^9$ tonnes CO_2 e per year or 18% of the global anthropogenic methane emissions were due landfills and waste water treatment processes in 2004-2005. Developing countries have been estimated to account for about 29% of the global emissions, but this share is expected to increase rapidly reaching about 64% by 2030 (Monni et al., 2006). This is predicted due to growth in population and affluence, expansion of waste collection services and improved landfill management (i.e. change from dumps to sanitary landfills, most without landfill gas collection systems). Therefore, it is very important to understand these emissions and to focus mitigation initiatives in the waste sector on developing countries and their disposal facilities.

Methane emissions from landfills are routinely calculated and very rarely measured directly. The decomposing of waste in landfills and the resultant methane (and landfill gas) is calculated with the help of models which are used to summarize the very complex chemical and biological decomposition. Several such models (some simple, others complex), with different orders of kinetics have been developed, namely zero-order, first-order and second order models, as well as some more complex models (Kamalan et al., 2011). The most popular ones have been the first order models and overviews and formulae for the most used first order models (GasSim, LandGEM, TNO, Belgium, Afvalzorg, EPER and Scholl Canyon) have been presented by Kamalan et al. (2011) and by Thompson et al. (2009). In particular, a variation of the Scholl Canyon model has been used by the IPCC in their 1996 and 2006 guidelines on how to calculate methane from landfills (IPCC, 1996 and IPCC, 2006). These guidelines have been aimed at estimating GHG inventories from waste at national level and to report them in an internationally agreed methodology. However, the IPCC calculation model has also been used at a regional, municipal and landfill site scale (Weitz et al., 2008 and Wangyao et al., 2009) in developing countries.

There are a variety of factors which influence the generation of landfill gas and methane (Komilis et al., 1999), however, the three key factors for methane generation models for a landfill site are: the amount of waste disposed of in the landfill since commissioning, the degradable organic fraction of that waste and the decay rate (of each organic fraction and as a whole) (Thompson et al., 2009). Since for many developing countries records on the amounts of waste landfilled at a particular site, and in general, have not been always kept and the composition of the waste was not always known, in many cases estimations and extrapolations had to be used. Most notably, the IPCC guidelines (2006) established a method that can be applied to all countries/regions and provided default values (e.g. the regional generation rates as presented in Table 2.1), estimates and calculation methods to overcome lack of historical data. However, these estimates introduced higher uncertainty in the final results and countries with poor waste management data (which are mostly developing countries) have the highest uncertainties in their calculations. In the IPCC 2006 Guidelines (IPCC, 2006) uncertainties for global emissions from waste for developed countries with good data availability have been estimated to be 10-30% and for developing countries that do not have annual data it was estimated at 60% and above. These uncertainties have been traced back to the lack of data with regard to the amount and composition of the waste, but also to assumptions that have to be used (decomposition rates, methane generation rates, oxidation rates, capturing efficiency, etc). In addition Lou and Nair (2009) highlighted that, in practice, the overall rate of emission for landfill gas can be also influenced by operational interventions, like waste compaction, leachate recirculation or aerobic landfilling and theoretically these factors should also be taken into consideration when modeling methane generation.

In general, the main criticism with regard to methane prediction models is their lack of accuracy and validation (Bogner and Matthews, 2003; Thompson et al. 2009, etc.). Thompson et al. (2009), for example, showed that although the four first order models investigated (LandGEM, TNO, Belgium and Scholl Canyon) *"have the same basic components with slight differences, their outputs vary considerably"*. This highlights why methane generation models have to be validated (i.e. predicted methane has to be compared with methane recovery data) and only a few, individual studies carried through this kind of investigation (e.g. Thompson et al., 2009 and Spokas et al., 2006). One of the more accurate methods to validate methane prediction models at landfill sites is the carbon balance approach (Spokas et al., 2006). This approach takes into account that the methane generated can be oxidized, recovered and stored within the landfill site. It can also migrate and only the remaining amounts are emitted into the atmosphere. Each component of the carbon balance can be quantified, modeled, engineered and optimized in order to reduce the amounts of methane emitted to the atmosphere. In particular, the capturing of landfill gas (for flaring and energy recovery), the oxidation of methane by using compost landfill cover, the pretreatment of waste and aerobic landfilling have been investigated as GHG mitigation strategies for landfills and have been covered in a review by Lou and Nair (2009). The implementation of such technologies in developing countries have been hampered by a series of factors, including lack of finance and capacity, however, with the implementation of the CDM projects a positive trend can be observed.

In developing countries the CDM projects sparked the use of the methane prediction models, carbon balances and their validation, because these processes were the prerequisites to predict the technical and financial feasibility of individual projects. Bogner et al. (2008) estimated that more than $105*10^6$ tonnes CO_2 e per year are recovered from landfills world-wide. Methane recovery was initiated in the waste sector in 1975 and it is implemented (mandatory and voluntary) in many developed countries and in particular the USA. In developing countries it is also used and it became more financially feasible after the opening of the CDM process. Monni et al. (2006) predicted that in 2008 about $30*10^6$ CO_2 e were recovered globally due to CDM projects in developing countries (representing about 28% of the global methane recovery). This figure does not include the offset energy due to the utilization of methane. Scenario modeling by the same authors showed the potential contribution that developing countries can play in the future, if gas capturing schemes continue to be implemented post 2012 (when Kyoto and CDM end) and an overall 15% global methane recovery rate is to be achieved (considered optimistic).

Unfortunately, the African continent lags behind and has the smallest numbers of CDM projects (in general and for the waste sector) registered or applying for registration (CDM Statistics, 2010; Couth and Trois, 2010 and 2011). Even so, out of the CDM work some regional validation will be possible for African countries and some model parameters could be customized in the future.

2.6.1.1 Calculated overall GHG emissions from landfill sites

The overall calculations for GHG from landfill sites involve a life cycle assessment approach which extended beyond the use of methane (and landfill gas) generation models and carbon balances, and included emissions from transport, the materials and energy used for constructing the site, the operation of the site, etc. Gentil et al. (2009) proposed the upstream-operating-downstream conceptual framework for accounting and reporting of GHG from waste, which takes into account direct and indirect emissions and savings. In this context, and taking a life cycle approach, Barton et al. (2008) compared GHG emissions for a series of generalized waste disposal scenarios applicable for developing countries. They have concluded that sanitary landfills with no landfill gas capture will have the highest GHG emissions (1.2 t CO_2 e/t of waste), followed by open dumpsites (0.74 t CO_2 e/t of waste), sanitary landfills with gas collection and flaring (0.19 t CO_2 e/t of waste) and sanitary landfills with gas collection and electricity generation (0.09 t CO_2 e/t of waste). These figures are for wet waste. In their sensitivity analyses, the waste composition proved to be a critical factor and with waste composition moving towards a more developed country composition, the amount of greenhouse gases increased in both open dump and landfill without gas collection scenarios (mainly due to the increase in paper and textile). Therefore, this study underlined the increased emissions that developing countries will have if the waste is dumped or disposed as practiced currently.

In a generic study, Manfredi et al. (2009) calculated emission factors for landfills in developed countries (Europe) based on a lower biogenic carbon content of the waste. Although the boundaries were different, the results are in the same range of those presented by Barton et al. (2008) for developing countries. Open dumping (included only for comparison purposes) accounted for about 1 t CO_2 e per tonne of waste, sanitary landfills with gas collection and energy recovery accounted for 0.3 t CO_2 e per tonne of waste and low-organic-carbon landfills (for Europe, but not applicable for the majority of developing countries) for 0.07 t CO_2 e per tonne of waste. These figures refer to wet waste. No other scenarios were investigated. The study concluded that energy recovery is important and the actual amounts of GHG savings depend on what the generated energy substitutes. Another important conclusion was that stored biogenic carbon in landfills should be also considered, since it proves important in the European context. Manfredi et al. (2009) quantified greenhouse savings of 132 to 185 kg CO_2 e per tonne of wet waste in the European landfills. These conclusions are also valid for developing countries and they need further research for regional quantification.

Focussing on landfill sites alone, the international guidelines took into account methane emissions in different types of landfill sites including dumps, and a methane correction factor (MCF) is used for calculations and incorporated in the overall formulae for methane emissions (UNFCCC, 2008). The UNFCCC methodological tool used to determine methane emissions from disposal of waste at a solid waste disposal site (UNFCCC, 2008) which can be used with the IPCC first order decay model (IPCC, 2006) uses the following values for the methane correction factor for landfill sites:

- *1.0 for anaerobic managed solid waste disposal sites. These must have controlled placement of waste (i.e. waste directed to specific deposition areas, a degree of control of scavenging and a degree of control of fires) and will include at least one of the following: (i) cover material; (ii) mechanical compacting; or (iii) leveling of the waste;*
- *0.5 for semi-aerobic managed solid waste disposal sites. These must have controlled placement of waste and will include all of the following structures for introducing air to the waste layer: (i) permeable cover material; (ii) leachate drainage system; (iii) regulating pondage; and (iv) gas ventilation system;*

- *0.8 for unmanaged solid waste disposal sites – deep and/or with high water table. This comprises all solid waste disposal sites (SWDS) not meeting the criteria of managed SWDS and which have depths of greater than or equal to 5 meters and/or high water table at near ground level. The latter situation corresponds to filling inland water, such as pond, river or wetland, by waste;*
- *0.4 for unmanaged-shallow solid waste disposal sites. This comprises all solid waste disposal sites not meeting the criteria of managed SWDS and which have depths of less than 5 metres (UNFCCC, 2008)*.

2.6.2 GHG emissions from waste incineration

It is estimated that over $130*10^6$ tonnes of waste are incinerated per year in over 600 plants world-wide (Bogner et al., 2008) and many developed countries derive significant benefits in terms of fuel replacement and energy from waste. However, controlled incineration as a method for waste disposal for municipal waste is not wide-spread in the developing world due to higher costs and unsuitable waste composition (high organic fraction, high moisture percentages and lower calorific value) (Barton et al., 2008). GHG emissions from incineration are considered small at around $40*10^6$ tonnes CO_2 e per year (less than one tenth of the emissions from landfills) (Bogner et al., 2008). When accounting greenhouse gases from incineration the biogenic carbon is not included, being considered neutral. Therefore, only fossil carbon (from plastics, synthetic textiles, etc) is accounted and reported (IPCC, 2006 and Astrup et al., 2009). In addition, the 2006 IPCC methodology specifies that for national inventories GHG emissions from the incineration of municipal waste are to be reported in the Waste sector if there is no energy recovery and under the Energy sector if there is energy recovery (IPCC, 2006).

Astrup et al. (2009) quantified greenhouse gases from incineration and co-combustion in the European context using the upstream-operation-downstream approach. They reported emissions from operations as 347-371 kg of CO_2 e per tonne of waste for incineration with energy recovery and 735-803 kg of CO_2 e per tonne of waste for co-combustion from municipal waste. However, there are savings of about -480 to -1373 kg of CO_2 e per tonne of waste from incineration with energy recovery and -181 to -2607 kg of CO_2 e per tonne of waste from co-combustion. In the developing world, for Taipei City, Chen and Lin (2008) report a saving of 222 kg of CO_2 e (or 0.06 MTCE) per tonne of waste from incineration with energy recovery. This value was calculated by a similar life cycle methodology to the upstream-operation-downstream approach followed by Astrup et al. (2009), however, the boundaries and the approach (generic vs case study) in the two studies are different. For the Taipei study the operational and transport emissions have been already subtracted from the savings. The generic European study and the figures calculated for Taipei show that in most cases (in developed and developing countries) GHG savings are larger than operational emissions and that incineration technologies can have substantial benefits in terms of energy generation and fuel replacement.

Another form of incineration which is practiced on a much larger scale in developing countries is the uncontrolled, open burning of waste. In developed countries this practice is prohibited. This kind of combustion is practiced at small scale (back-yard) and at larger scale (in landfills) and can be spontaneous (e.g. in poorly managed landfill sites due to methane) or set deliberately in order to reduce the volume of waste. Most of the studies on the topic of open burning of municipal waste are investigating the emissions of toxic compounds and the potential health risks they pose (e.g. Lemieux et al., 2004). There are no reported values for GHG emissions due to this activity at municipal level, however, the 2006 IPCC guideline (Chapter 5) contains a methodology for calculating GHG emissions from the open burning of waste to be applied for national inventories (IPCC, 2006).

2.6.3 Greenhouse gases from composting

Composting is used in both developed and developing countries as a way of dealing with the biodegradable fraction of their municipal waste (Bogner et al., 2008). Composting offers real advantages not only by reducing the volumes of waste but also by recycling nutrients and organic matter and improving soils. Since the decomposition process is aerobic, composting also generates less greenhouse gases as compared to landfilling. In Europe alone there are about 2000 composting facilities for household organic waste (Boldrin et al., 2009) and there is a successful policy to divert organic wastes from landfilling into composting. In developing countries composting should provide a viable alternative, because of the high biodegradable fraction of the waste. However, many of the large-scale, earlier initiatives involving composting in these countries (including Africa) failed and the smaller, decentralised operations seem to be currently more successful (Cofie et al., 2009). The CDM mechanism can also be used for composting in developing countries and a methodology has been developed for large scale projects (AM0025) and for small scale projects (AMS-III.F).

Currently there are two successful CDM municipal solid waste (MSW) composting projects in Africa. The first is in Cairo (Egypt) and involves mechanical and manual sorting of dry waste, followed by the shredding and turned windrow composting of the wet waste (UNFCCC PDD, 2007). The second CDM composting project is in Lagos (Nigeria) (UNFCCC PDD, 2009), and involves the shredding of unloaded waste followed by windrow composting. In addition, in Khartoum (Sudan) a composting plant is at planning stages (Tawfig et al. 2009).

In Europe and Australia composting has been used within the mechanical biological treatment technologies (MTB) for the stabilization of the organic fraction of the waste. The use of this compost is highly regulated in the OECD countries (UNEP, 2010). In some developing countries (e.g. Pudong, China) MTB technologies are also starting to be used (Hong et al., 2006).

Composting contributes to the release of GHG emissions, but more important it also saves such emissions and the actual amounts depend on a series of factors including waste composition (i.e. organic fraction), composting technologies, use of gas cleaning (i.e. for enclosed systems) and the actual use for the final product (Boldrin et al., 2009). Lou and Nair (2009) summarise the main theoretical and practical studies quantifying emissions from the composting process itself, showing that theoretical estimates (0.284-0.323 t of CO_2 e per tonne of waste) usually overestimate real, measured emissions (0.183-0.932 t of CO_2 e per tonne of waste). However, since the process is aerobic and emissions are of biogenic origin they are not accounted for and the emissions which really matter in the case of composting are the operational emissions. Lou and Nair (2009) also showed that greenhouse gas emissions are usually lower for windrows composting as compared to aerobic in-vessel composting due to lower energy requirements. They also underline that, although studies have shown that methane and nitrous oxide are produced during composting, they are usually not included in GHG accounting for this process.

Boldrin et al. (2009) present more detailed and extensive overall GHG emissions for composting and they do include methane and nitrous oxide emissions in an accounting methodology which uses the upstream-operating-downstream approach. They show that emissions can vary between -0.900 (net savings) to 0.300 (net load) t of CO_2 e per tonne of wet waste composted. They covered four composting technologies, namely open composting (windrow, static pile, mat), enclosed composting (channel and cell and aerated pile), reactor composting (tunnel reactor, box and container and rotating drum) and home composting. They show that the upstream contributions are very little, the operation contributions are moderate and the main burdens and savings in terms of greenhouse gases come from the use of the compost and what the compost substitutes. Other published results for developed countries (-183 kg of CO_2 e per tonne of waste for the USA (EPA, 2006) and between -32 to -58 kg of CO_2 e per tonne of waste for Europe (Smith et al., 2001) fall within the range calculated by Boldrin et al. (2009). These figures refer to wet waste. Quantification of greenhouse gases has been done only for the use of compost on land (replacing synthetic fertilizer) and peat substitution. Substitution of fertilizer is estimated to save about 8 kg of CO_2 e per tonne of wet waste composted and applied to land and substitution of peat will save about -4 and -81 kg of CO_2 e per tonne of wet waste composted (Boldrin, 2009). Farrell and Jones (2009) show that compost can be used for many other applications, most notably for remediation. The GHG emissions/savings from these uses are not quantified in the literature.

There are also a series of uncertainties with regard to GHG emissions from composting due to lack of scientific consensus (e.g. nitrous oxide emissions during compost use) or lack of actual data (i.e. for what the compost will be used). In particular, estimates and calculations for developing countries will have even more uncertainties included. It is considered that there is a paucity of specific data for composting in developing countries and studies done for these countries use data from literature (Boldrin et al., 2009). Calculations performed by Barton et al. (2008) for composting in developing countries in general, confirm this (i.e. the use of literature data) and the results in this case show that composting as process is carbon neutral. An almost zero effect of the composting process is also reported by Zhao et al. (2009) for Tianjin (China). They also show that, if kitchen waste (which represents about 57% of the waste) from this municipality is composted instead of landfilled, a 24 % reduction in GHG emissions can be achieved. This illustrates the potential of composting in terms of greenhouse savings for municipalities in developing countries which have waste with a high organic content suitable for composting.

2.6.4 Greenhouse gases from anaerobic digestion

Anaerobic digestion has been defined as the anaerobic decomposition of organic wastes which produces biogas (methane and carbon dioxide) and biosolids (digestate) and as a waste management technology is practiced by both developing and developed countries. Developed countries, especially in Europe, have focused on high-tech plants and developing countries historically used low-tech smaller plants/reactors in which manure and other organic wastes were digested. However, in the last few decades a series of new local initiatives (plants/technologies) were introduced in developing countries, but not all were sustainable in the long term (Müller, 2007). Other (e.g. BARC (Mumbai), ARTI (Pune)) low tech anaerobic digestion technologies developed in these countries show real potential (Müller, 2007). Unfortunately, African countries are lagging behind with only a few experimental initiatives in this area (Müller, 2007). Most notably is the introduction in Tanzania of the ARTI system developed in India (Voegeli et al., 2009).

For the European context, Møller et al. (2009) assessed the overall GHG emissions from anaerobic digestion of source-separated municipal solid waste using the upstream-operating-downstream framework for the accounting of these gases. Their results showed that overall emissions from anaerobic digestion vary between a saving of -375 to a burden of 111 kg of CO_2 e per tonne of wet waste. The emissions from specific types of AD facilities varied between savings of -95 to -4 kg of CO_2 e per tonne of wet waste. They showed that, if an AD facility has high biogas production, substitutes CO_2-heavy electricity and exports heat, the savings could be substantial. However, if there are low methane yields, in connection with upgrading of biogas to vehicle fuel and high emissions of nitrous oxide from digestate, then a net burden will result (Møller et al., 2009). Smith et al. (2001) calculated slightly higher overall GHG savings for anaerobic digestion in the same geographical context. Their results ranged from -246 to -51 kg of CO_2 e per tonne of wet waste.

For developing countries in general, Barton et al. (2008) using a life cycle assessment estimated theoretical savings of – 210 kg of CO_2 e per tonne of wet waste due to the use of anaerobic digestion. More specifically for Tianjin (China), Zhao et al. (2009) calculated that anaerobic digestion is almost carbon neutral. In another case study for Phuket (Thailand) Liamsanguan and Gheewala (2008) calculated a saving of -30 kg of CO_2 e per tonne of wet waste treated by anaerobic digestion. Results from these studies in the developing world are within the range presented by Møller et al. (2009) for Europe. These differences in emissions are due to a variety of reason linked to the overall set-up and efficiency of the overall system. Biogas yields and the nature of the energy that this biogas use avoids play the most important role and will determine the ultimate savings.

Similar to composting, the overall quantification of greenhouse gases from anaerobic digestion has a high degree of uncertainty associated with it and Møller et al. (2009) identified the key parameters influencing emissions from anaerobic digestion. These are (1) substitution of energy or natural gas by biogas, (2) nitrous oxide emissions from digestate in soil, (3) fugitive methane emissions at the plant, (4) unburned methane during combustion, (5) carbon bound in soils and (6) fertilizer substitution. Some of these parameters are hard to quantify in developed countries, but even more so in developing countries, and even if case specific data is available, a certain degree of uncertainty will still persist (Møller et al., 2009).

2.6.5 Greenhouse gases from recycling

There is agreement in the literature (e.g. EPA, 2006; Smith et al., 2001; Christensen et al., 2009) that recycling of fractions of municipal waste offers some of the highest benefits with regard to GHG savings from waste. Recycling is practiced by both developed and developing countries and differences (legal, social, economic and technical) have been noted in the literature (e.g. van Beukering and van den Bergh, 2006 and Uiterkamp et al., 2011). Recycling is a complex waste management issue which is beyond the scope of this paper, however, in terms of GHG emissions it presents definite advantages for all municipalities in all countries.

Different greenhouse savings have been reported for different recycled materials and Table 2.3 presents a summary from the published literature. As it can be seen from this table the greenhouse savings from recycling vary for each of the materials considered. However, the most common themes when investigating these variations are energy and the different variations in the downstream substitution in the use of the recycled material. As can be seen from Table 2.3 there is agreement that recycled aluminum has the highest potential savings in term of greenhouse gases, followed by steel, plastics, paper and glass, which show some of the lowest savings.

Table 2-3: Greenhouse gas savings from recycling different fractions of municipal waste (expressed as tonne of CO2 e per metric tonne of waste unless otherwise specified)

Waste fraction/Material	Smith et al. (2001) for Europe	EPA (2006)* for USA	Other authors (European context)
Paper - mixed	-0.60	-3.19 (-0.96)	-390 to -4.40 (Merrild et al., 2009)
Plastics – HDPE	-0.49	-1.26 (-0.38)	-1.27 to -0.99 (Astrup et al., 2009)
Plastics - PET	-1.76	-1.40 (-0.42)	
Glass	-0.25	-0.27 (-0.08)	-0.50 to -1.50 (Larsen et al., 2009b)
Ferrous metal (steel)	-1.48	-1.63 (-0.49)	-0.56 to -2.36 (Damgaard et al., 2009)
Aluminium	-9.07	-12.31 (-3.70)	-5.04 to -19.34 (Damgaard et al., 2009)

*original values (EPA, 2006) expressed in MTCE/ton (US) are presented in brackets

The savings due to recycling are expected to be higher in the developed world and in particular in those countries which rely on coal as a predominant source of energy. South Africa is such a country, as are India and China. Therefore, to quantify savings from waste recycling more precisely, country specific and even region specific saving factors should be calculated also for developing countries in a similar fashion as those for Europe and the USA (see Table 2.3). So far there are no such recycling factors for the developing countries, and in the few studies from the developing world quantifying savings from recycling the EPA (2006) factors from Table 2.3 are used (e.g. Chen and Lin, 2008; Friedrich and Trois, 2008 and Chintan, 2009). In general, there is a paucity of life cycle assessment studies in the developing world as compared with developed countries, and these studies are used in the quantification of greenhouse gases from more complex waste management systems. The literature review conducted by the authors for this study alone yielded about 40 research articles, peer reviewed publications and reports for developed countries and only about 10 for their developing counterparts. In particular, Africa seems to lack such studies.

Two other issues pertinent to recycling in developing countries and the emissions of greenhouse gases are the export/import of recyclables to and from developing countries and the role of the informal sector in recycling. The export and import of recyclable and recycled materials to and from developing countries is becoming important in a globalised world. For example, in South Africa in 2009, 73 tonnes of recycled paper were imported and 17 tonnes of recycled paper were exported (PRASA, 2010) with the country being a small player in the recyclables market. In general, recycled materials are exported to developing countries from the developed world. An example is the UK where during 2007 $4.7*10^6$ tonnes of paper and $0.5*10^6$ tonnes of plastics recycled in this country were exported to China (WRAP, 2008) and therefore the transport of these recyclables over long distances might reduce the greenhouse savings substantially. However, this seems to be case specific. A study done by WRAP (2008), for the export of recyclables from the UK to China, showed that in general less than a third of the CO_2 emissions are due to transport. These emissions drop to less than 10 % if taking into account the fact that a large number of ships return empty from the UK to China. This conclusion might not be generalized and it depends on the mode of transportation and the waste transported (Salhofer et al., 2007) and more research is needed in this area. Increased quantities of electronic waste are exported to African countries (Schmidt, 2006) where the waste is further processed for recycling and it causes an array of environmental and health problems (Nnorom and Osibanjo, 2008 and Robinson, 2009).

The informal recycling sector (i.e. waste pickers who salvage recyclables in the waste management system) in developing countries plays an important role in reducing greenhouse gases as shown by Chintan (2009) for India. For Delhi alone about $962*10^3$ tonnes CO_2 e was saved by the informal sector recycling, which achieved very high recovery rates (e.g. mixed paper 95%, mixed plastic and metals 70% and glass 75%). These informal GHG savings compare favorably with other formal initiatives (CDM projects for waste-to-energy and composting) being more than three times greater (Chintan, 2009). In addition to GHG savings, the informal recycling sector supplied an income for about 15 million waste pickers in 2007 alone and brought other advantages to the formal waste management system at local level (e.g. reduced volumes of waste, savings on costs for collection, transport and disposal, extended life of a landfill) (Wilson et al., 2006 and Medina, 2008). However, these marginalized groups are not supported by authorities, lack access to finances (e.g. carbon trading scheme) and are in conflict with formal reduction projects (e.g. access to recyclables is reduced in the case of waste to energy projects). A similar situation with regard to informal recycling is reported for the African countries (Ball et al., 2007) including South Africa (Oelofse and Strydom, 2010), however, there is a lack of information on the quantities recycled by the informal sector as well as on the other advantages due to this activity. Furthermore, the South African recent legislation (Waste Bill - Act 59 of 2008) does not recognize the role of waste pickers in municipal waste management.

2.7 Conclusions

This review paper compared the GHG emissions from different municipal waste management processes in developing and developed countries, with particular emphasis on the African continent and South Africa. What sets developing countries apart are the different motivational factors for GHG accounting and reporting. Developing countries do not have a mandatory obligation to report GHG and there are less data and information for waste management in general and in particular for the quantification of greenhouse gases. In the absence of such data, a variety of assumptions have to be used, which affects the accuracy of calculations and makes validation of results a very challenging process. One example of such an assumption is the waste generation rate for African countries (IPCC, 2006) which currently seems to be over-estimated. In addition, the GHG emissions from waste management in developing countries are predicted to increase exponentially. Therefore, more attention has to be paid to how these emissions arise, are accounted/calculated and reported for waste management processes in the municipalities of developing countries.

When investigating GHG emissions from individual processes there is agreement on the magnitude of the emissions expected from each process (generation of waste, collection and transport, disposal and recycling). Recycling brings about the highest savings in terms of GHG, followed by composting and incineration with energy recovery. The disposal of waste in landfills has some of the highest GHG emissions. In particular, in developing countries these emissions are dominating due to the methane released by dumpsites and landfills. If these are upgraded to sanitary landfills these emissions will continue increasing, unless the methane is captured and either flared or used for electricity generation. The CDM projects have made some in-roads with regard to the waste to energy projects, however, the African continent lags behind. The GHG emissions from transport and collection are lower in developing countries due to inadequate provision of these services, in particular in African cities which have some of the lowest collection rates.

The investigation of GHG emissions from individual waste management processes, which make up a waste management system, show that the few values (e.g. GHG emissions for landfilling or transportation) calculated for developing countries are within the range reported for developed countries. However, one has to be critical of these results, because there are no calculations done for the elements/processes found only in developing countries (e.g. non-motorized transport of waste). A direct comparison of GHG emissions from waste management in different municipalities should be undertaken only at process level. At systems level such comparisons should be undertaken with care, because the determining waste management factors (e.g. waste composition, collection rates, waste management process, etc.) are different and so could be the accounting methodology used. Therefore, there is a need to develop a common approach applicable for developed and developing countries for the accounting of greenhouse gases from waste management at municipal level and individual processes should be the foundation blocks.

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Chapter 3 Greenhouse Gases Accounting and Reporting for Waste Management - A South African Perspective

3.1 Abstract

This paper investigates how greenhouse gases are accounted and reported in the waste sector in South Africa. Developing countries (including South Africa) do not have binding emission reduction targets, but many of them publish different greenhouse gas emissions data which have been accounted and reported in different ways. Results show that for South Africa, inventories at national and municipal level are the most important tools in the process of accounting and reporting greenhouse gases from waste. For the development of these inventories international initiatives were important catalysts at national and municipal levels, and assisted in developing local expertise, resulting in increased output quality. However, discrepancies in the methodology used to account greenhouse gases from waste between inventories still remain a concern. This is a challenging issue for developing countries, especially African ones, since higher accuracy methods are more data intensive. Analysis of the South African inventories shows that results from the recent inventories cannot be compared with older ones due to the use of different accounting methodologies. More recently the use of CDM (Clean Development Mechanism) procedures in Africa, geared towards direct measurements of greenhouse gases from landfill sites, has increased and resulted in an improvement of the quality of greenhouse gas inventories at municipal level.

3.2 Introduction

South Africa is a developing country with a population of about 49 million (Statistics South Africa, 2009). About 58% of the population in South Africa is urbanised (DEAT, 2007). The country underwent a rapid urban growth in the late 1980s and 1990s and this process continues. It is expected that about 68% of the population will live in cities and towns by 2015 (GCIS, 2009). Increased and rapid urbanisation has placed significant pressures on service delivery in urban areas, including housing and associated waste management. About 25% of the urban population is living in slums with various degrees of service provision. Service delivery in the country, including waste management, is the responsibility of local government and large disparities exist. The large metropolitan municipalities provide a complete service in terms of waste management, including collection and appropriate disposal. However, many smaller municipalities in rural settings lack the capacity to deliver any waste management services at all. In general, it is estimated that about 50% of the population of South Africa is not receiving regular waste collection services (DEAT, 2007).

Currently the majority of waste in South Africa is disposed in landfills. There are about 475 landfills, which have been granted permits in terms of the old waste legislation/guidelines, and there is evidence of about 760 sites (legal and illegal) that are operating without a permit (DEAT, 2007). There are also many smaller, unrecorded sites in rural areas. This has direct implications for the generation and accounting of greenhouse gases from waste management, because landfills are generating important quantities of landfill gas, which contains methane. Methane is a potent greenhouse gas, having a global warming potential 21 (Lilieveld et al., 1998) to 33 (Shindell et al., 2009) times higher than carbon dioxide for a 100-year time frame. Globally, it is evaluated that methane emissions from landfills and wastewater account for 90% of the greenhouse gases from these sectors and contribute to about 18% to the total anthropogenic methane emissions (Bogner et al., 2008).

The last decade brought about a proliferation of accounting and reporting tools for greenhouse gases to be applied on different scales (nations, regions, cities, companies) by different entities (government institutions, private companies and even individuals). For waste management there are three approaches used. Firstly, the Greenhouse Gases Inventory approach, which accounts for these gases at a large (macro) scale (e.g. municipality, region, country). Secondly, the Clean Development Mechanism (CDM) approach which is project specific (micro scale). Thirdly, greenhouse gases from waste are evaluated using the Life Cycle Assessment (LCA) or more specifically the carbon footprinting methodology. Life cycle assessments and carbon footprinting have been applied at macro (entire waste management system) and micro (project and/or activity) scale. However, to date LCAs and carbon footprinting for greenhouse gas emissions from waste have not been employed in South Africa and although used in some developing countries (e.g. Chen and Lin, 2007) they are not as widely employed as in the developed world.

This research aims to investigate how greenhouse gas accounting and reporting from waste is undertaken in South Africa, with reference to one of its largest cities. The focus is not on technical greenhouse gas prediction models but more on the accounting and reporting process at municipal and national levels and the use of different tools in this process.

3.3 Methodology

Exploratory research on the accounting and reporting of greenhouse gases was sparked by the recent release of the South African Inventory for 2000 (GHG Inventory South Africa, 2009). Greenhouse gases emissions from waste are indeed accounted and reported at national and municipal levels, but these separate processes have not been researched in detail. In particular, the development and use of the inventories at local level for waste, as well as the interaction between inventories and CDMs are areas needing investigation. To research this interaction the eThekwini Municipality has been used as a case study because it houses the first CDM landfill gas-to-electricity project in Africa and produced greenhouse gas inventories for 2001-2002 and for 2005-2006.

In the first stage of the research a desktop study was undertaken to obtain background information on national inventories, the methodology employed, uncertainty and the assumptions made for South Africa. Municipal inventories (international and local ones) were also consulted, in order to assess the way greenhouse gas emissions from waste were incorporated and what methodology was used. In the initial stage of the research, municipal environmental departments in South Africa were contacted and copies of the greenhouse gas inventories for the City of Johannesburg, City of Cape Town and the eThekwini Municipality were obtained.

In a second stage a series of interviews followed by questionnaires were conducted with the municipal employees in charge of the inventories and with the consultants that produced the inventory reports. The aim of this stage was to collect detailed, municipality-specific data on the models used for the prediction of greenhouse gases from landfill sites and to research how the figures generated from these models have been incorporated in the municipal inventories. Initially, the questionnaires were emailed. Due to a low response and some clarifications needed, the emails were followed-up with face-to-face or telephonic interviews. Questions were open ended and for environmental departments/consultants focussed around the initiatives in compiling inventories at municipal level and how the process was managed and evolved. For waste entities' employees, the questions were focussed around data collection for models (waste type and quantities), the models used, validation of the models through field trials and other aspects of the CDMs in their area.

3.4 National greenhouse gas inventories and the case of South Africa

Developing countries are not bound by the Kyoto Protocol (UNFCCC, 2008), but most of them (including South Africa) are producing national greenhouse gas inventories as part of the United Nations Framework Convention on Climate Change (UNFCCC), following the International Panel on Climate Change (IPCC) Guidelines (IPCC, 2006). These guidelines provide an international framework for the measuring, accounting, verifying and reporting of greenhouse gases for the main sectors, including waste (IPCC, 2006) and the methodology recommended is based on a tiered approach (with three tiers) depending on data availability. Tier one is the most basic approach with less data requirements and less accuracy of calculations. Tier two and three approaches are more complex, need more data and are generally more accurate (Kennedy et al., 2009a). For the waste sector, the calculation and accounting methodology has changed. In the previous (1996) IPCC guidelines a yield model was used, in which it was assumed that all the greenhouse gases from the waste produced in the year of the inventory are released in that year, although in reality the decomposition of waste in landfills and the release of gases take decades. The 2006 IPCC guidelines use a kinetic, first order of decay model which takes into account that greenhouse gases due to the waste landfilled in a particular year are released over several years in different amounts.

The flexibility (the tiered approach) in methodology and data required allows a unified, documented approach and each country "*regardless of experience and resources*" (IPCC, 2006) should produce reliable estimates. However, this flexibility is also a weakness because in the absence of real data the amount of assumptions it allows might introduce a high degree of developing countries and particularly in Africa. The uncertainties range from 30% for countries which collect waste generation data and have periodic sampling on waste composition (like

South Africa), to more than a factor of two (60 %) for countries with poor quality data (IPCC, 2006).

The challenges of producing greenhouse gas inventories in developing countries have been noted in the literature (Price, 1996; Ravindranath and Sathaye, 2002). They have broadly been classified as related to data (availability, access to and accuracy of) and nationally relevant emissions factors and to financial, institutional, political and organisational issues. Lack of historical data, problems with institutional capacity and the need for inter-agency co-operation were some of the issues highlighted. The paucity of information on waste management and the resultant greenhouse gas emissions can be attributed to the prevailing problems that developing countries (especially African ones) are facing with regard to waste management systems (Ball, 2006; Henry et al., 2006; Manga et al., 2008; Matete and Trois, 2008; Ogawa, 2008). This is only partially valid in the case of South Africa and its greenhouse gas inventories.

The first South African national GHG Inventory was prepared in 1998, using 1990 data and a second one followed in 2004 based on the 1994 data (GHG Inventory South Africa, 2009). In May 2009 a third inventory was released based on data from 2000. The three inventories produced were commissioned by the Department of Environmental Affairs and Tourism (DEAT) and were prepared by a consortium of research organisations (the Energy Research Centre – University of Cape Town and the Centre for Scientific and Industrial Research (CSIR)). The work on the GHG inventory from waste was undertaken by the CSIR (GHG Inventory South Africa, 2009).

There is a large disparity between the results of the three national inventories with regard to the contribution of waste to the emission of greenhouse gases. Overall results for 2000 show a decrease of 38.2 % from 1990 and 42.8% from 1994. This is in spite of growing population and the amount of waste generated per capita. It highlights two critical points, namely the importance of the models used in predicting methane generation from waste in landfill sites and the danger of comparing generic inventory tables (as presented in the summary of the national 2000 inventory (GHG Inventory South Africa, 2009)) without investigating the methodology behind them. The first two inventories used the IPCC 1996 guideline (which stipulates the gas yield model) and the third and most recent inventory uses the IPCC 2006 guideline (with the first order of decay model which was employed at tier two level).

In the 2000 inventory a series of assumptions had to be made in applying the calculation methodology as stipulated by the IPCC 2006 guidelines for the waste sector. In the inventory, only managed landfill sites were taken into account. Due to lack of data, unmanaged landfill sites and industrial sites were excluded. Another important set of data that was lacking with regard to the calculations performed, were statistical data (from the 1990s and earlier) on waste quantities and the composition of waste, which are critical parameters for greenhouse gas generation. A series of default values (e.g. the methane generation rate and the waste decay half-lives) for bulk waste under dry temperate conditions were used as per IPCC 2006 Guidelines. The errors due to assumptions ranged from 15 to 20 % for 2001 for which real data existed and which was just outside the inventory period. This margin of error was considered acceptable and within the limits of the 30 % for countries that collect waste generation data on a regular basis (GHG Inventory South Africa, 2009). In the literature similar margins of error (30-40%) are reported (Lim et al., 1999 and Rypdal and Winiwater, 2001) for developed countries with better data management systems. As lack of data has been identified to be the most important limiting factor, an investigation of the collection of data within the South African waste system was undertaken.

South Africa started collecting data for waste from the late 1990s and a few voluntary waste information systems have been initiated in the country. However, the majority have failed (Godfrey and Nahman, 2007). At national level the only one that operates to some degree is the national waste information system under the umbrella of the Department of Environmental Affairs (DEA). It needs upgrading to meet the new requirements of the Waste Act (Republic of South Africa, 2008) and a discussion document as part of preparing the new National Waste Management Strategy was released in August 2009. This process has important implications for the modelling of GHG from waste because it will allow much more precise inputs and calculations for methane generation. However, there are serious challenges in the implementation of such a system and regulations requiring a more detailed waste stream analysis have been promulgated recently, putting poorer municipalities under financial/capacity stress (Purnell, 2009).

3.5 Municipal greenhouse gas inventories and the case of the eThekwini Municipality

In the literature, as presented by Kennedy et al. (2009a), there are five different methodologies to calculate municipal greenhouse gases from waste disposed in landfills. These methods are: (1) scaling from national inventories, (2) the gas yield approach, (3) the EPA's WARM model (a simplified LCA based model), (4) the first order of decay model advocated by the IPCC (based on a Canyon Scholl approach) and (5) measurement from waste in-place. The same authors recognise that for municipal inventories "*the determination of GHG emissions associated with waste is where the greatest discrepancies in methodology are apparent*" (Kennedy et al., 2009a). Dodman (2009) also observes that the greenhouse gas emissions due to waste are varying significantly between global cities. For New York, due to landfill gas recovery (assumed 75%) and carbon sequestrations, the net emissions from waste are negative (City of New York, 2007). However, other cities like Atlanta (5%), Sao Paulo (23.6 %), Barcelona (24 %), Rio de Janeiro (36.5%) report different percentages in their inventories (as shown in brackets), which are attributed to waste. Dodman (2009) further concludes that "*these variations are likely to be due not only to different patterns of consumption and waste generation but also to differences in the management of waste and differences in accounting mechanisms – variations that are impossible to assess in the absence of a standardised urban framework for conducting emissions inventories*".

Historically, there have been attempts towards a common approach for local and regional greenhouse gas inventories. Most notably, the International Council for Local Environmental Initiatives (ICELI) most recently referred to as Local Governments for Sustainability and their Cities for Climate Protection campaign managed to launch valuable initiatives, not only in developed countries, but also in a few developing countries (ICELI, 2009) like: India, Thailand, Argentina, Brazil and South Africa. Another initiative targeting European cities and regions is GRIP (Greenhouse Gas Regional Inventory Protocol) (Carney et al., 2009). In addition, there are wider initiatives like ISO 14064, or the protocol developed and promoted by the World Resources Institute and the World Business Council for Sustainable Development, which give principles for the accounting and reporting process for greenhouse gases. However, these standardisation initiatives do not address, in particular, emissions from waste, and they are either allowing different approaches to be used or they favour one particular approach. In terms of developing countries, especially African ones, this has particular implications. Since most of the inventory studies are funded from international sources and there is (or was – as in the case of South Africa) little local experience, the application of different approaches is perpetuated and different methods are used depending on affiliations, not necessarily on suitability.

3.5.1 The eThekwini Municipality case study

The eThekwini Municipality is one of the largest municipalities in South Africa and is situated on the eastern coast of the country. It covers an area of about 2300 square kilometres and has a population of approximately 3.1 million. The core city of the municipality is Durban, which has the largest South African port on the Indian Ocean. Inland, Durban is surrounded by other urban nodes, as well as other more sparsely populated peri-urban areas. In the mid 1990s the boundaries of the municipality were extended incorporating many previously segregated informal settlements. This move instantly added a population of about 1,250,000 people to the municipality and posed significant challenges in terms of service delivery (housing, water, electricity, waste removal, etc.)

Durban Solid Waste (DSW) is the municipal waste disposal unit and operates 3 active sanitary landfill sites (Bisasar Road, Mariannhill and Buffelsdraai landfill sites), 23 recycling and garden refuse drop-off centres, 6 major transfer stations, 3 landfill-gas to electricity plants and 3 leachate plants (Durban Solid Waste, 2009). In terms of waste collection and management, the levels of service differ between different areas of the municipality. In the formal areas there is a regular collection service provided (usually once a week) for domestic waste (which is collected in black bags) and garden refuse (collected in blue bags) for which households are charged. For the informal settlement areas limited collection services are provided at defined drop-off points free of charge. DSW also provides street cleansing and verge maintenance for all the roads of the municipalities.

In addition to DSW, which is a municipal entity, in the eThekwini municipality there are two private companies (Wasteman and Enviroserv), which are involved with collection and disposal of waste. They cater mainly for industrial customers but do accept limited amounts of municipal (household) waste. Each operates a sanitary landfill site designed, classified and licensed to accept low hazardous waste. Table 3.1 summarises all five landfill sites (municipal and private) which are located in the municipality and which are the major sources of methane.

Table 3-1: Landfill sites in the eThekwini Municipality

Landfill	Tons received/day	Ownership	SAn Classification
Bisasar Road	3880	DSW	G:L:B+
Mariannhill	690	DSW	G:L:B+
Buffelsdraai	140	DSW	G:L:B+
Shongweni	About 480	Enviroserv	H:h
Bulbul Drive	About 860	Wasteman	H:h

Classification: G=general waste, L=large, B+=leachate production, H:h=low hazardous waste

Greenhouse gas accounting and reporting (including those from waste) for the eThekwini Municipality were initiated in 2002 (and later in 2006) as a result of an international initiative through the involvement of the ICLEI. In general, in the South African context the ICLEI played an important role through their Cities for Climate Protection campaign. It was launched in South Africa in 2001 and initially assisted nine local municipalities (including the eThekwini Municipality) to produce greenhouse gas inventories. Recently (2009) there were 13 South African municipalities registered as members of the ICLEI (ICLEI, 2009). For the South African municipalities the CCP campaign provided some financial help, an emission inventory guideline and a software calculator specifically developed for municipalities. Initially, for local greenhouse gas inventories, the software developed by Torrie Smith & Associates was used. In 2004, as an extension on the ICLEI work done in developed countries, the HEAT (Harmonised Air Emission Analysis Tool) software was customised for a series of developing countries including South Africa (van Staden, 2009).

The results for the 2001-2002 inventory show that waste disposal was responsible for 187 377 tons CO_2e, representing 18% of the municipal greenhouse gas emission. To perform calculations DSW provided information about the amount of waste and the different types of waste collected and disposed during the financial year 2001-2002. However, the DSW waste classification did not coincide with the waste classification provided by the Torries Smith & Associates software. To reconcile these differences, various assumptions had to be made regarding the biodegradable portions of the waste stream (GHG Emission Survey, 2002). The margin of error due to these assumptions has not been calculated. Most notably, one small municipal landfill (La Mercy) and two privately operated low hazardous landfill sites, which also disposed considerable amounts of municipal waste, were not included in this inventory. The two largest municipal landfill sites (Bisasar Road and Mariannhill) had gas collection and flaring installed at that time and the reduction of emissions due to this process has been quantified.

The 2006 municipal inventory was more inclusive, calculating and reporting GHG emissions due to local government activities as well as from community emissions. The HEAT software was used and, with regard to the waste sector, it presents a choice with regard to the methodology to be used (the yield method vs. the first order of decay model), as well as the possibility to enter greenhouse gas emissions calculated by another model altogether. It also has a calculator for savings from capturing landfill gas as well as changes in waste practice (HEAT, 2009). The results show that only about 5% (1 118 061 tons CO_2e) of the total emissions are due to local government greenhouse gases, as opposed to 18% calculated in the previous inventory. These emissions include the three municipal landfill sites (Bisasar Road, Marianhill and La Mercy), which produced 195 888 tons CO_2e. The emissions calculated for the two privately owned landfill sites amounted to 103 656 tons CO_2e and were included under "community emissions". The amount of methane released from the three municipal landfill sites was calculated by Enviros-UK using the GasSim Model, while the calculations for the two privately owned landfills were done by a local consultancy, using historical data and the same assumptions were applied, although the model was somewhat simplified. The data provided by the private companies to the consultants performing the greenhouse gas calculations have not been externally verified, whereas the GasSim-generated data was externally audited and became the framework for the CDM projects.

3.5.2. The eThekwini Municipality CDM project

The Clean Development Mechanism (CDM) and the associated Certified Emissions Reductions (CER) provide a very defined and controlled way of greenhouse gas accounting and reporting for project specific waste emissions in the developing world (Jewaskiewitz et al., 2008). Worldwide, CDM projects are managed by the CDM Executive Board by means of registering the projects. There are precise steps/stages in this process and the roles of all players involved are well specified. Third party validation is used through the Designated Operational Entities (DOEs), who are independent auditors, which ensure that the project results in real, measurable, and long-term emission reductions. However, due to the expertise and data required to engage in this process, statistics show that very few projects from Africa regarding waste management have succeeded (CDM Statistics, 2009). Moreover, there is uncertainty about the future of this system at the end of the Kyoto period and beyond 2012, and this might have resulted in an evident reduction in applications for registration of new CDM projects (Cornish et al., 2008).

The eThekwini Municipality and Durban Solid Waste initiated the first landfill gas to electricity project to be registered under the CDM system in Africa in 2002 (UNFCCC reference number 0545). The main benefit in terms of GHG accounting was the use of precise measurements instead of theoretical calculations on the amount of methane emitted by landfill sites. The CDM methodology specifies how baseline emissions (in absence of a project) should be calculated and monitored, ensuring quality control and eliminating previous calculation errors and data uncertainties. A series of other factors are specified, most important being the recording frequency, the proportion of data to be monitored, how data will be aggregated and archived (and for how long). The methodology is aimed to ensure quality control and assurance for the monitoring process and includes regular maintenance of equipment. In the eThekwini, landfill sites collection and flaring of landfill gas was already undertaken prior to the engagement with the CDM system, however, detailed predictions and accounting of methane generation, as well as accurate estimation of efficiency of landfill gas collection were never required before, nor was a standard for maintenance and data collection.

3.6 Discussion

The results for the waste sector from the two municipal greenhouse gas inventories (2002 and 2006) can not be directly compared due to the use of two different methodologies and there was no attempt to perform recalculations on the older inventory data. The quality of the 2006 Greenhouse Gas Inventory for waste managed by DSW was much improved due to the use of more accurate landfill gas prediction models sparked by the work required in the preliminary stages of the CDMs projects. Therefore, the CDM scheme had a very positive impact on municipal greenhouse gas accounting for waste and this trend is expected to continue in a new inventory, as data from operating the landfill gas to energy plants and from the field trials are becoming available. The 2006 inventory is more inclusive by accounting for the emissions from the two privately owned landfill sites and the closed municipal La Mercy landfill site. Therefore, locally the trend with regard to the process of accounting is to improve the quality of municipal waste greenhouse gas inventories in time. In this process, local expertise was and is developed as more municipal employees (from DSW, but also from the municipal environmental department) and private consultancy firms are involved in the process of producing the inventories. None the less, several limitations are identified.

One of the most important limitations in the 2006 local Greenhouse Gas Inventory is the fact that different standards are applied in the accounting of greenhouse gases for DSW and the two private waste management companies active locally. The data on DSW operations is being monitored and verified by international external auditors; however, the private companies are not being independently monitored with regard to the data supplied to consultants for greenhouse gas modelling and calculations. Another, even more important, limitation in the 2006 inventory is the splitting of the emissions due to waste management. The emissions from DSW are included under municipal emissions and those from the two private waste companies are included under community emissions. A more holistic and integrated approach to waste management and to the accounting of greenhouse gases from waste should be taken at local level. In the absence of such an approach it would be harder to quantify the overall consequences of any intervention (e.g. recycling initiative, education campaign, etc.).

An integrated approach is also needed when looking at the role of recycling and other waste minimisation initiatives and the greenhouse gas savings they bring about, particularly because this topic is not well researched at national and municipal level in South Africa. Currently, these savings may or may not be incorporated in greenhouse gas inventories due to waste. In the South African national inventory, which was based on waste generation rates per capita, these savings were not included. At municipal level they might be included depending on the methodology used. In the 2006 eThekwini Municipality inventory savings from waste minimisation were excluded for the municipal (DSW) landfill sites, as well as for the private landfill sites calculations. In general, these savings are not incorporated in any methodology based on generic waste composition data and/or scaling the local greenhouse gas emissions down from a national inventory. The authors recommend that the boundaries for municipal inventories should be extended to account for waste minimisation activities in order to present more accurate and realistic greenhouse gas emission figures for the waste sector.

The theory and the practice around the accounting and reporting of greenhouse gases are constantly being improved and there are two recent theoretical developments which have potential consequences for the waste sector. The first is a continuation of the trend towards standardising municipal inventories and a generally applicable methodology was proposed by Kennedy et al. (2009b). The second development is the debate around the conceptual structure of municipal inventories (i.e. production vs. consumption inventories) and the implications thereof.

Since the major problem around the accounting and reporting of greenhouse gases from waste is the lack of consistency in the accounting methods used, new inventory methodologies suggested for cities have been targeted and Kennedy et al. (2009b) propose a “new” method for calculating greenhouse gases due to waste management as part of a standardised method for accounting city-wide greenhouse gases. This method is, in fact, a derivation of the gas yield method and arguably it might be more appropriate for African cities, since it is much simpler and requires less data (less than the first tier approach of the 2006 IPCC Guidelines). If applied consistently, it will also allow comparisons between different municipalities and inventories. On the other hand, this method will perpetuate the use of different inventory methodologies (gas yield vs. first order of decay models) for waste, but it might offer a solution for cases in which historical waste data is unobtainable and higher accuracy models cannot be applied. However, for the eThekwini Municipality and the other African municipalities involved with landfill gas to energy CDM initiatives, the application of only the method proposed by Kennedy et al. (2009b) will be a step backwards, since it is based on calculations (with their inherent assumptions and margins of errors) and not actual measurements. It also does not have any mechanism for data validation and quality assurance.

Another issue is the debate around "production vs. consumption" types of inventories (Peters and Hertwich, 2008). At the heart of this debate is how the boundaries are set in allocating greenhouse gas emissions, because "*emissions can be attributed either to the spatial location of actual release or to the spatial location that generated activity that led to the actual release*" (VandeWeghe and Kennedy, 2007). Production-based municipal inventories are the traditional ones, which include the greenhouse gas emissions in a geographically delimited area (city, municipality, region), not taking into account that the products manufactured in this area may be consumed in other places (Dodman, 2009), posing considerable disadvantages for exporting countries (e.g. China and even to some degree South Africa). A consumption- based inventory will take into account the greenhouse gas emissions due to the consumption of products and services taking place in an area regardless where these products/services originate. These two different ways of allocating greenhouse gas emissions have particular implications for the potential emissions due to waste in South Africa. No specific accounting methodology has been proposed to address this issue, but theoretically for consumption-based inventories, all the greenhouse gas emissions due to the waste produced by the manufacturing of products for export should be subtracted and the emissions due to the waste produced by the products/services imported should be added. At the moment, we consider such a complex accounting system, which requires intensive data collection and sharing, not applicable in the South African waste management context.

3.7 Conclusion

The results and analysis presented in this paper show that, in the accounting and reporting of greenhouse gases from waste in South Africa, the most important role is played by calculated inventories, followed in recent years by the use of CDM procedures which allow direct measurements of greenhouse gases (mainly methane) from landfill sites. Although not bound by the Kyoto Protocol and with no national legislation forcing disclosure, South Africa has produced three national inventories (for the years 1990, 1994 and 2000) and most of the largest municipalities in the country have produced municipal greenhouse gas inventories too. One of the most important limitations of these inventories is the change in the methodology used for calculating greenhouse gases from landfills – i.e. the replacement of the gas yield model by the first order of decay model. This change is in line with IPCC guidelines and should achieve more realistic results and higher overall accuracy; however, the calculated emissions of successive inventories produced by the different methodologies should not be compared.

For South Africa the results for the 1990 and 1994 inventories cannot be compared with those for 2000, as they show a decrease of greenhouse gases due to waste that is incorrect. In reality the emissions have increased substantially. Establishing a trend requires recalculations by using the same methodology. However, these interpretations should be specified in the national inventory and particular in summary tables and not left open to misinterpretation.

When it comes to municipal greenhouse gas inventories from waste, the methodological choice is even wider and five different ways for greenhouse gas accounting and reporting are presented in the literature. In South Africa, the methodology used was influenced by the affiliation to international initiatives, which promoted the production of the municipal inventories. The first municipal inventories (which included emissions from waste) were produced at the beginning of the 2000s under the initiative of the CCP (Cities for Climate Protection) campaign launched through the ICLEI (International Council for Local Environmental Initiatives). For the eThekwini Municipality there were two successive inventories (for the years 2001-2002 and 2005-2006) prepared and a third one is in the initial stages. For the waste sector some conclusions are pertinent, namely: the quality of the inventories produced has increased due to the interaction with the CDM landfill gas to energy projects and due to the development of local capacity to undertake municipal greenhouse gas inventories. The inventory published in 2006 uses more realistic greenhouse gas accounting methods and is more complete in terms of including all municipal landfill sites (owned by the municipality and by private companies). This trend towards improvement is expected to continue as real measurement data obtained through the CDM projects is becoming available and will be incorporated in the new inventory. Therefore, CDM projects have the indirect benefit of increasing the quality of local greenhouse gas inventories due to waste and setting a standard for how greenhouse gases should be accounted and monitored at landfill sites. The recommendation for improvement is to spread these standards and to develop a locally relevant system for waste data (in accordance with proposed national initiatives) and the resultant greenhouse accounting system for the landfill sites which are not part of the CDM system. The CDM projects and their extensive data quality insurance should be regarded as best practice. Therefore, the CDM projects achieved major advances with regard to greenhouse gas accounting and recording in a country where disclosure is not mandatory.

In summary, although facing a series of limitations, the preparation of greenhouse inventories in South Africa is considered a valuable exercise in understanding inventories in general and those addressing the waste sector in particular. The need for more reliable data for waste generation and waste composition also adds to the need for reliable information in the South African waste management context and to the development of a national waste quantification and information system. Although there are challenges with regards to the implementation of such a system in all South African municipalities, the final result, even with partial implementation, will lead to a better input for further national and municipal waste inventories and will set a clear example for other developing/emerging countries.

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Chapter 4 Greenhouse Gas Emission Factors

4.1 GHG Emission Factors Developed for the Collection, Transport and Landfilling of Municipal Waste in South African Municipalities

4.1.1. Abstract

Greenhouse gas (GHG) emission factors are used with increased frequency for the accounting and reporting of GHG from waste management. However, these factors have been calculated for developed countries of the Northern Hemisphere and are lacking for developing countries. This paper shows how such factors have been developed for the collection, transport and landfilling of municipal waste in South Africa. As such it presents a model on how international results and methodology can be adapted and used to calculate country-specific GHG emission factors from waste. For the collection and transport of municipal waste in South Africa, the average diesel consumption is around 5 dm^3 (litres) per tonne of wet waste and the associated GHG emissions are about 15 kg CO_2 equivalents (CO_2 e). Depending on the type of landfill, the GHG emissions from the landfilling of waste have been calculated to range from -145 to 1016 kg CO_2 e per tonne of wet waste, when taking into account carbon storage, and from 441 to 2532 kg CO_2 e per tonne of wet waste, when carbon storage is left out. The highest emission factor per unit of wet waste is for landfill sites without landfill gas collection and these are the dominant waste disposal facilities in South Africa. However, cash strapped municipalities in Africa and the developing world will not be able to significantly upgrade these sites and reduce their GHG burdens if there is no equivalent replacement of the Clean Development Mechanism (CDM) resulting from the Kyoto agreement. Other low cost avenues need to be investigated to suit local conditions, in particular landfill covers which enhance methane oxidation.

4.1.2. Introduction

The waste management sector has been identified as one of the sources of greenhouse gas (GHG) emissions, being responsible for about 5% of global emissions (Bogner et al., 2007) and for 4.3% of the South African GHG emissions (GHG Inventory, 2010). The waste management processes not only generate GHG (e.g. collection, transport and landfilling), but also have the potential to save such emissions by displacing use of virgin materials which have much higher emissions associated with their production (ISWA, 2009 and Scheutz et al., 2009). Usually, performing a carbon balance for alternative waste management processes, will show where net savings and emissions occur. These net savings and emissions for a process, expressed per unit (usually tonne, ton or Mg) of waste (wet, dry or both) have been termed as GHG emission factors and are used extensively in the quantification of GHG from waste (e.g. Obermoser et al., 2009; Gentil et al., 2009 and Levis and Barlaz, 2011). The development of such factors for countries and regions enables the calculation of GHG in a faster and transparent manner. This in turn enables wider and more sophisticated applications for GHG accounting in the waste sector, in most cases supported by the development of specialist software, and allows for better waste related decision-making in the context of climate change (United States Environmental Protection Agency – U.S. EPA, 2006). The aim of this study is to present the GHG emission factors that have been developed for the municipal waste management processes currently used in the South African waste sector. The results of the study are presented in two parts. Part 1 (this paper) presents the GHG emission factors developed for collection, transport and landfilling of municipal waste. Part 2 (future paper) will present these factors for recycling (glass, plastics, paper and metals - namely Al and Fe) and composting.

The quantification of greenhouse gases has to be seen in a wider context in which a waste management system operates within and interacts with other systems of a country or region. The quality of carbon balances and life cycle assessment studies for waste (and the resultant emission factors) depend on the availability and quality of data regarding waste management, but also on data on the energy system of that country or region. South Africa, as well as many other developing countries (e.g. China and India), relies heavily on coal for energy generation. Therefore, any waste management process which will save (e.g. source reduction and recycling) or replace (e.g. landfill gas to electricity) this fossil fuel intensive energy will have positive effects in terms of overall GHG emission reductions. Some of these aspects are fundamental to the Clean Development Mechanism (CDM) projects for Non-Annex 1 countries (i.e. landfill gas to electricity). However, other aspects (i.e. recycling and transport of waste as part of an integrated waste management system) have not been well investigated locally in terms of associated GHG emissions (Friedrich and Trois, 2010). In addition, because of different capacities of local South African authorities, these investigations are not always possible, especially in the smaller municipalities in rural areas, which lack resources and know-how to perform even the basic waste management functions, let alone GHG accounting. Even in the large municipalities which do record waste data, the process of quantifying and accounting GHG from waste management is generally targeting landfill sites emissions (due to the CDM requirements) and not the entire waste management system. By ignoring the emissions from other processes in the waste management system (e.g. transportation, recycling, waste minimisation) and by focussing mainly on processes which bring revenue (i.e. landfills with CDM projects) the overall optimisation of these systems may be jeopardised and important avenues for GHG reductions from waste ignored. Therefore, the aim of this research was to develop South African emission factors for waste management in order to perform a better quantification of these emissions for all processes and a uniform application within the waste sector.

4.1.3. GHG emission factors for waste management and associated calculation models with reference to developing countries

In GHG accounting an emission factor is a term used to present the amount of GHGs released per unit of energy, mass, or volume (Merrild et al., 2009). Usually these emission factors are calculated by employing a life cycle assessment (LCA) approach, in a simplified or more sophisticated form. This implies that in the calculation of an emission factor, emissions per given unit have been accounted and summarised over an entire life cycle of a product/service/activity or over a part of it, depending on how the boundaries of these studies have been set. Therefore, it is not surprising that the use and development of emission factors mirrors the development of LCA approaches. As LCA initiatives matured and became standardised (e.g. ISO 1046 and PAS 2025) and their usage increased, the sophistication and coverage (by processes, as well as by geographical spread) of emission factors also increased, leading to better and more detailed accounting of GHG emissions, including from the waste sector. In particular, the development of carbon footprinting as a form of simplified LCA is closely related to the increased use and development of GHG emission factors.

For the waste sector GHG emission factors have been developed and incorporated in many LCA-based accounting models developed for different audiences as shown in reviews by Del Borghi et al. (2009), Gentil et al. (2010) and Björklund et al. (2011). These accounting models for waste management systems, as well as more recent studies which published in-depth factors (ranges and calculations) for individual waste management processes are summarised in Table 4.1 and Table 4.2. These studies show that there is a variety of applications for emission factors and associated calculation tools. These range from specialist applications and tools targeted to waste managers and LCA professionals to applications and tools which are designed for household on-line calculations and the general public.

Table 4-1: Current tools using emission factors for waste management systems (adapted from Del Borghi et al., 2009; Gentil et al., 2010 and Björklund et al., 2011)

Ownership/Authors	Associated model and/or calculation tool	Country of origin
EU, 1995 and 2001	LCA-IWM (1 and 2)	EU
Environment Canada, 1999	IWM Canada	Canada
International consortium formed by institutions in the USA and Europe (Camobreco et al., 1999)	Life cycle inventory	USA EU
Solano et al., 2002 and Thorneloe et al., 2007	MSW-DST	USA
USA EPA, 2006	WARM	USA
Tanaka et al., 2004 in Gentil et al., 2010	SSWMSS	Japan
Nordic Council of Ministers, 2007	Own tool (organic waste emissions)	Nordic countries (Denmark, Sweden, Norway, Finland)
Environmental Agency of Wales	WRATE	UK
Swedish research institutes (SLU, IVL, JTI, KTH)	ORWARE	Sweden
Ecobilan, 1999	WISARD	UK
Technical University of Denmark (Kirkeby et al., 2007)	EASEWASTE	Denmark
Anderson et al. (2010)	KONSTA (on-line calculator for households), MARTTI (material flow accounting) and PETRA (waste benchmarking)	Finland

Most of the tools and publications (see Table 1 and 2) deriving and using emission factors have been developed in the Northern Hemisphere. For developing countries, this evolution is double sided; on the one hand it is positive because it sets the methodological framework and tested pathways for developing emission factors, and reference values for different processes (including waste management) are made available. This is an aid with regard to GHG accounting, especially for countries which lack data and capacity to develop their own emission factors. On the other hand it can have negative consequences, because the existence and use of emission factors out of the context in which they have been developed and without understanding how these factors have been calculated and derived, might introduce large margins of error and give false results when applied in the developing world. Therefore, there is a need to develop local GHG emission factors which reflect as closely as possible the reality in a particular country or region. A number of studies (e.g. Merrild et al., 2008; Vergara et al., 2011) have shown that significant variability in emission factors derived from LCA-based approaches can occur as a result of using different assumptions and assigning different boundary conditions for establishing criteria. Some of these factors are geographically conditioned (e.g. electricity generation and the resultant GHG emissions) while others are independent of location (e.g. the technological performance of equipment at landfill sites which translates into efficiency factors and GHG emissions).

Table 4-2: Academic publication calculating emission factors for waste management processes

Waste management (WM) process	Authors	Country of origin
a) Collection and Transport	Chen and Lin, 2008 Larsen et al., 2009a Eisted et al., 2009 Nguyen and Wilson 2010	Taiwan, Denmark, EU Canada
b) Landfilling	Manfredi et al., 2009 Levis and Barlaz, 2011	EU USA
c) Composting	Boldrin et al., 2009	EU
d) Combustion	Larsen and Astrup, 2011 Obermoser et al., 2009 Kaplan et al., 2009	EU EU USA
e) Recycling – Paper Plastics Metals Glass	Merrild et al., 2009 Astrup et al., 2009 Damgaard et al., 2009 Larsen et al., 2009b	All EU
f) Anaerobic digestion	Møller et al.2009	EU
g) Waste-to-energy plants	Obermoser et al. (2009) Kaplan et al., 2009	EU USA

4.1.4. The methodology involved in the development of GHG emission factors

The development of emission factors is based on a carbon balance approach and such balances are usually performed by conducting a series of simplified or streamlined life cycle assessments for each waste management process investigated. In general the methodological framework followed in this study is conceptually similar to that one used by a series of international studies (as presented in Table 4.1). However, a reduced number of waste management processes have been investigated, because of their relevance to South Africa (e.g. no municipal waste is incinerated in the country and anaerobic digestion is still under investigation) and the availability of data in the South African context.

The development of emission factors for waste has to be seen in the wider context of waste management in the country, and in particular the availability of data was a limiting factor. Only 68.9 % of municipalities in the country collect some form of waste data and 74.6% of these municipalities collect only data for the waste which reaches the landfills (Godfrey, 2008). Therefore, it is considered that "*while fairly extensive waste data is now collected by metropolitan municipalities; it is clear that accurate municipal waste data, for the majority of municipalities in South Africa, is not available*" (Republic of South Africa, Department of Environmental Affairs – RSA DEA, 2012; pp.3). In this context, data on the collection and transport of waste and landfilling, as well as on waste composition were obtained from published sources, as well as through interviews, site visits and direct observations. Table 4.3 presents the types of data used in this paper, as well as data sources and observations regarding data quality.

In this paper the reference unit, used for data collection, for calculations and for presenting the results, is 1 tonne (1 tonne = 1000 kg = 1 Mg) of solid, wet, general waste. Although the South African definition of general waste includes building and demolition waste, in this paper this waste was excluded since it is considered inert with regard to GHG emissions. The general waste referred to in this study is generated by households and businesses in South African municipalities and does not include other categories of waste (e.g. medical waste or hazardous waste) which need special handling and disposal, as stipulated by local legislation and regulations.

Table 4-3: Types of data used, sources of data and observations regarding quality of data used

Type of data	**Source**	**Observations**
Consumption of diesel and petrol for waste collection and transport	Records from four different South African municipalities	One municipality had records only for one year, restricting statistical analysis and one municipality had missing data sets (i.e. for a few months). Calculated averages had to be used to substitute missing data.
Waste composition	Different published waste compositions from different municipalities were used and are presented in Table 4-6.	Most of these compositions originate from integrated waste management plans, which were generated by different consultancies. Therefore, the quality of data is different and there was no data quality check possible (e.g. no of samples, period sampled, etc.).
Degradable organic carbon (DOC) content of waste fractions	IPCC, 2006	Based on international literature. No local, more detailed data were available. DOC values show optimal degradation rates which might overestimate the actual degradation in landfills in particular in the arid parts of South Africa where degradation might be limited due to lack of moisture.
Generation and distribution of electricity	Eskom - South Africa's monopoly electricity generator and distributer	Country specific, South African electricity mix.

4.1.4.1 The development of a GHG emission factor for municipal waste collection and transport

Two approaches are possible when developing a collection and transport GHG emission factor for municipal waste. The first one is based on the use of general emission factors for common vehicles. Such transport emission factors have been developed in the literature and are incorporated in all mainstream LCA databases. These databases usually present fuel consumption and different emissions (CO_2, NO_X, SO_X, particulates, etc.) per unit of payload for different types of vehicles. In addition, some of these factors have been calculated to take into account different levels of technologies (e.g. EURO 0, 1, 2, 3, 4 and 5) and different road conditions (country roads vs. highways), which influence emissions. To use these emission factors, it is necessary to estimate average trip lengths and average loads for a chosen vehicle. In the first pioneering studies (e.g. Smith et al., 2001 and U.S. EPA, 2006) which calculated GHG emission factors for the collection and transport of waste this approach was used. These studies tended to underestimate fuel consumption and associated emissions because waste collection vehicles operate differently during the collection process as compared to normal transport. They frequently stop and go, idle during loading and use hydraulic compaction presses, resulting in higher fuel consumption and higher emissions. Therefore, in more recent literature, there are higher emissions factors for vehicles used for collection and transport of municipal waste. These emission factors have been derived through a second, more specific approach and are based on measured and/or specifically modelled fuel consumption of waste collection vehicles as opposed to other common vehicles carrying the same payload.

In this paper, an emission factor relevant to South Africa was calculated for the collection and transport of municipal waste by collecting data on fuel consumption from four different South African municipalities. A variety of vehicles are used in the collection and transport of municipal waste in South Africa and there are different topographical conditions in the different cities. The overall fuel consumption will account for such variability; however, direct fuel consumption is not usually recorded by municipalities. Godfrey (2008) showed that only 33.8% of municipalities in the country collected information on waste transportation and associated collection. In this study, fuel consumption was calculated based on the expenditure (in a financial year for each municipality) used for fuel for waste and the average price of that fuel for that year. To obtain an average per unit of waste, the total annual fuel consumption was divided by the amount of waste delivered to landfill sites by municipal vehicles, in the same year. A variable amount of waste (5-20%) for each municipality was collected and delivered to landfill sites by private contractors working for the municipalities. It was assumed that these collectors used similar vehicles and had the same fuel efficiency as municipal vehicles.

The GHG emissions resulting from the diesel combusted are CO_2, CH_4, and N_2O, however the amounts of CH_4 and N_2O are negligible relative to CO_2, but were included in this study. GHG emissions were calculated by using the emission factor for diesel, which in this study was taken as 2700 g CO_2 e per dm^3 (litre) of diesel (Fruergaard et al., 2009). This emission factor is slightly higher than the one used in other local studies (2686 g CO_2 e per litre of diesel – used in the Durban Industry Climate Change Project), but the difference is considered negligible. Such a methodological approach was followed by other international studies published in the field as illustrated in Table 4.4.

Although the methods of calculating and/or measuring diesel consumption and the GHG emission factors from the diesel vary between the different studies, there is agreement that the consumption of diesel and the associated GHG emissions for collection are higher in the low-density and rural fringes of the cities in developed countries as opposed to collection from the high density urban areas of these cities (Larsen et al., 2009 and Nguyen and Wilson, 2010). Based on this observation, four different South African municipalities have been targeted for data collection. Two of the South African municipalities are large, metropolitan municipalities (eThekwini Municipality and City of Cape Town), one is a middle-sized town (uMsunduzi Municipality) and one is a small municipality with a large rural component (Hibiscus Coast Municipality).

Table 4-4: Fuel consumption for the collection and transport of municipal waste in urban areas

Municipality/location	Diesel consumption* (dm^3 $tonne^{-1}$)**	Diesel emission factor (g GHG dm^{-3} diesel)	GHG emissions due to diesel (kg CO_2 e $tonne^{-1}$)	Reference	Observations
Taipei (Taiwan) - overall	5.9	2747 (derived through calculations)	16.4	Chen and Lin, 2008	Collection and transport calculated are based on overall diesel consumption. Overall density is 9700 people km^{-2}
Aarhus (Denmark) – high density urban (apartment buildings and the city centre) – medium density urban (single family houses incl. those in small towns) – rural areas	 1.6 – 3.6 1.4-5.7 6.3-10.1	2629 (Euro 4 and 5 standards)	 4.2– 9.5 3.7-15 16.6-26.6	Larsen et al., 2009a	Collection only, based on diesel measurements. No quantitative figures for densities for study area were presented.
Hamilton (Ontario, Canada) – high density urban – low density urban	 3.2 ± 0.4 14.6 ± 1.4	2730	 8.9±1.1 40.3±3.9	Nguyen and Wilson, 2010	Collection and transport only to transfer station. Based on measurements and calculations. Density defined for linear distance: Low density < 25 houses km^{-1} High density > 40 houses km^{-1}

*$1dm^3$ = 1 L

** mass refers to wet waste

4.1.4.2. Background information and the choice of different landfilling options

In the first stage, in the development of emission factors for landfills and dumps, information was reviewed pertaining to the status quo situation in the country. Based on this information the types of landfills (including dumps) and the assumptions associated with calculations for each type of landfilling were decided upon and are presented in the following sections.

It was estimated that about 59 million tonnes of general waste (including construction and demolition waste) were generated in the country in 2011 and of these about 5.8 million tonnes were recycled (RSA DEA, 2012). Waste generation rates have been reported for different income levels, namely 0.41 kg/capita/day for low income households, 0,74 kg/capita/day for middle income households and 1.29 kg/capita/day for high income households (RSA DEAT, 2006). Based on this rates and the proportion of different income levels in the general population an overall generation rate of 0.52 kg/capita/day has been calculated for the country. However, although widely used in the waste sector, these figures should be viewed with caution since they are based on only 3 studies from 1998, 2004 and 2005. In recent years the City of Cape Town and the City of Johannesburg reported overall generation rates above the 2 kg/capita/day benchmark. For example, in Cape Town from July 2006 to June 2007 a rate of 2.32 kg/capita/day was reported. This rate declined to 1.15 kg/capita/day in the period from July 2008 to June 2009 due to the economic down turn and some waste minimisation initiatives (City of Cape Town, 2009).

Landfilling is the main disposal method in South Africa and there are 1203 landfill sites recorded in the country. Of these, 524 (43.6%) are permitted in terms of existing legislation (RSA DEA, 2011). However, even for permitted landfill sites there is little or no information on compliance to permit conditions (i.e. air, water and soil pollution control) (Godfrey, 2008). For this study it was assumed that all permitted sites have some engineering features (e.g. access control, regular cover and some form of lining). Of the permitted landfill sites currently (2012) only 5 are sites with landfill gas (LFG) collection and flaring and only 3 sites collect and generate electricity from LFG. In addition to the recorded landfills, it is estimated that there were about 15000 unrecorded private and communal unregulated dump sites, situated mainly in rural areas (RSA DEAT, 2006). The Census 2011 Report (StatsSA, 2012) showed that 28.2 % of the households in South Africa used their own refuse dump for disposal and 1.9 % used a communal refuse dump. 5.4 % of households did not have access to any waste disposal facility and 0.9 % of households used other ways of disposal. Only 63.6 % of the households had their refuse collected and removed by local authorities and/or their private subcontractors. From the refuse collected and removed from households and businesses, some recyclables might be diverted, but the bulk of this waste will end up in a landfill. Using assumptions from the latest National Waste Information Baseline Report (RSA DEA, 2012) and assuming that all general commercial and industrial waste (including construction and demolition waste) is disposed in regulated landfills, it was calculated that about 25.4 million tonnes municipal waste is generated by households and not recycled. If the percentages from the Census 2011 Report (StatsSA, 2012) are used it results that for 2011 about 8 million tonnes municipal household waste ended up in dumps and about 45 million tonnes household and commercial waste in permitted, regulated landfills. Therefore, emission factors relevant to South Africa were calculated for landfill disposal in a dump, in a landfill site without landfill gas collection, in a landfill site with landfill gas collection and flaring and in a landfill site with gas collection and electricity generation.

4.1.4.3 GHG from landfill sites and dumps

The most important greenhouse gas emitted by landfill sites (including dumps) is methane, therefore, the calculation of emission factors for landfill sites relies on the amounts of biodegradable carbon in the waste. Overall GHG emissions due to the anaerobic decomposition of the waste are modelled without taking into account the time factor in the release of these emissions. Gentil et al. (2009, pp. 700) highlight that “the LCA methodology, with respect to GHG accounting, is rather different from the other accounting and reporting protocols, as its scope is to account for all potential emission and savings of waste management, regardless of the time”. Therefore, for landfills this approach has similarities to the gas yield model (IPCC, 1999) and differs from the other models which include the time factor (IPCC 2006 tier 2 and 3, Scholl Caynon model, GasSim, LandGem, etc.). Modelling emission factors for landfill sites relies on a carbon balance. Carbon enters the landfill site as organic material in the waste stream (i.e. food waste, garden waste, paper and wood). This carbon either exits the site as methane, carbon dioxide, volatile organic compounds (VOCs), or dissolved in leachate or is stored in the landfill (U.S. EPA, 2006).

Besides the methane produced by the anaerobic decomposition of waste in landfill sites, other GHG are emitted due to the construction (e.g. the preparation and lining of cells), operation (e.g. use of compactors which need fossil fuels) and decommissioning (e.g. maintenance and monitoring of closed sites). Scheutz et al. (2009) summarises these emissions as presented in Table 4.5. These GHG emissions were investigated in more detail for the South African situation and used to derive emission factors.

Table 4-5: GHG from landfilling (Source: Scheutz et al., 2009)

Upstream indirect emissions	Operation direct emissions	Downstream indirect emissions
Emissions of CO_2, CH_4, N_2O due to: Fuel production Electricity consumption Infrastructure (liners, soils)	Fugitive emissions of CH_4, traces of NMVOC, N_2O and halogen-containing gases. CO_2biogenic from waste decomposition Fuel combustion from machinery (CO_2, CH_4, N_2O, traces of CO and NMVOC) Emissions from leachate treatment plant (CO_2biogenic, CO_2, CH_4, N_2O)	Production of heat and electricity from CH_4 combustion substituting fossil energy (CO_2) Carbon bound in landfill (100 years).

4.1.4.4 The determination of carbon entering South African landfill sites

Different waste types contain different amounts of carbon in various forms. Some forms of carbon (e.g. cellulose) are easily degradable under landfill conditions while other forms (e.g. lignin) degrade slow or are non-degradable. Biogenic carbon is the carbon which is part of the short carbon cycle, i.e. the cycle in which carbon is moved between organisms and their surroundings through photosynthesis, growth, metabolism through the food chain, respiration and decay. Municipal waste contains biogenic carbon (e.g. food and garden waste) and fossil carbon (e.g. plastics). From the literature (Eleazar et al., 1997; Barlaz et al., 1998; IPCC, 2006; U.S. EPA, 2006 and Manfredi et al., 2009) typical ranges of biogenic carbon for the various waste fractions have been established and Manfredi et al. (2009) present a comprehensive summary. For the development of landfill site emission factors by the U.S. EPA (2006) a more precise approach was taken, whereby the total biogenic carbon was measured and given as a single figure for the waste types investigated (corrugated cardboard, newsprint, office paper, food discards, garden waste). Also an overall figure was measured for the 'average' municipal solid waste in the USA. These figures vary slightly if compared to the default values recommended by IPCC (2006), but for all materials they are within the IPCC range. Such precise measurements are not available for South Africa and, therefore, the IPCC default values have been preferred in this study when calculating the carbon entering the landfill sites.

Waste composition data is fragmented for South African municipalities with large municipalities recording and publishing this kind of information with varying frequencies, however, many of the small municipalities have no records at all. Taking into account the published information for different municipalities, a weighted average was calculated for the municipal waste composition representative for South Africa and is presented in Table 4.6. It was weighted according to the mass of waste collected and reaching the landfill sites. Where two waste compositions were found for the same city (but different years) the most recent data sets were taken for calculations. The waste composition for all municipalities were determined by analysing once-off or limited numbers of samples and no statistical analysis was possible in this regard. Different methodologies were used to determine these waste compositions, introducing some uncertainties in the average percentages calculated for each waste stream, however, in the absence of a unified methodology for waste stream analyses, these figures were considered acceptable.

Based on average South African waste composition (Table 4.6) and degradable organic carbon content (DOC) data for individual waste components from IPCC (2006), an average carbon input factor (CIF) was calculated and is presented in the last row of Table 4.7. A statistical analysis was performed by using Monte Carlo simulations. Average, maximum, minimum and ranges for 95% as well as 5 % cumulative distributions (giving 90% confidence intervals) have been calculated for all waste fractions and are presented in Table 4.7. In calculating these outputs the variability of the waste composition data (see Table 4.6) as well as the ranges for the DOC percentages of wet waste (IPCC, 2006) were taken into account simultaneously. The first step in the modelling process was to determine the waste category fraction distributions by the linear regression of the individual waste fraction amounts, against the total amount for all cities. These distributions were then put into the Monte Carlo simulation. The DOC factors for each category except 'other' were modelled with normal distributions, setting the mean to the central value, and taking the ranges to be 5% and 95% points of the distribution.

Table 4-6: Average South African municipal waste composition

Municipality	Reference	Waste Composition (% wet waste)							Amount of waste reaching landfills (tonnes)	Information on landfill sites
		Paper	Metals	Glass	Plastics	Food waste (Organic)	Garden waste (Green)	Other		
eThekwini Municipality (core city Durban) 3.6 million people	Trois et al., 2007	16	3	7	12	29	18	15	1654000 (measured for 2009)	4 permitted landfill sites (3 municipal and 1 private) engineered at high standards (2 of the municipal ones with LFG collection and electricity generation). LFG collection systems with electricity generation were developed as a CDM project.
City of Johannesburg 3.9 million people	Kwezi V3 Engineers, 2004 and Ball, 2001- both in Taiwo and Otieno, 2010	18.05	3	4.15	10.26	13.37	19.40	31.79	1492000 (measured for 2008)	5 permitted landfill sites (4 municipal and 1 private) engineered at high standards. 2 municipal landfill sites collect and flare LFG and there are plans to upgrade them to electricity generation. 3 municipal landfill sites (2 operational and 1 closed) are in different stages of planning LFG collection systems.
	Stotko, 2007	12	2	3	7	20	22	34		
City of Cape Town 3.5 million people	Stotko, 2007	15	2	4	5	31	20	23	1659400 (measured for 2011)	4 permitted landfill sites (3 municipal and 1 private) engineered at high standards. No LFG collection at the moment, but planned to start in 2014 as part of a potential CDM project.
	Council of the City of Cape Town, 2010*	19	5	8	13	47 together		8		
Nelson Mandela Municipality (core city Port Elizabeth) 1.1 million people	IWMP, 2010	18.8	3	6.3	14.9	46	4.3	6.6	619099 (measured for 2008)	3 permitted landfill sites with medium engineering (i.e. cells with liners and coverage).

uMsunduzi Municipality (core city Pietermaritzburg) 616 000 people	Trois and Jagath, 2011	21.1	4.2	6	7.6	32.1	0.7	28.3	169000 (measured for 2010)	1 permitted landfill with medium engineering (cells with liners and coverage)
Umdoni Local Municipality (core city Scottburgh) 62 290 people	IWMP, 2009*	12	4	5	7	45 together		27	31884 (measured for 2008)	1 permitted landfill site with basic engineering (i.e. covering of waste)
City of Potchefstroom 128 400 people	de Villiers, 2000*	24	4	8	10	51 together		3	647340 (measured for 2010)	1 permitted landfill site, medium engineering standards (cells using clay liner)
City of Mafikeng** 259 500 people	IWMP, 2011	4.2	3.97	14.39	7.14	Included with "Other"	9.74	60.51	52925 to 158775 (estimated for 2009 – no weighbridge)	1 permitted landfill site, non-compliance to permit specifications have been reported, low operational and engineering standards (i.e. free access)
Makana Municipality (core city Grahamstown) 75 000 people	IWMP, 2008	14	3	12	13	12	19	27	32986 (estimated for 2007- no weighbridges at 2 of the 3 landfill sites)	1 permitted landfill site 2 unpermitted landfill sites operated like dumps (i.e. waste is covered once a year), low engineering standards
Weighted Average	N/A	**18.2±0.65**	**3.9±0.35**	**6.9±0.54**	**12.1±0.45**	**26.0±2.6**	**18.2±1.14 (incl. wood) and 1.6±0.3** **(for wood)**	**14.7±3.35**	N/A	N/A

*for these three entries food waste and garden waste were classified together and therefore, in the calculation, a 50/50 proportion was assigned to each.

Table 4-7: Average DOC calculations and statistical outputs

Waste fraction	% in South African wet waste	DOC (%) for wet waste (IPCC, 2006)	DOC (%) range for wet waste (IPCC, 2006)	DOC outputs from the Monte Carlo simulation				
				Mean	Minimum	Maximum	5%	95%
Paper	18.2±0.65	40	36-45	72.78	53.65	94.14	64.10	81.44
Food	26.0±2.6	15	8-20	39.05	3.53	84.64	20.86	57.24
Garden waste (without wood)	16.6±1.1	20	18-22	33.90	26.22	41.95	30.56	37.24
Wood	1.6±0.3	43	39-46	7.05	5.83	8.35	6.49	7.62
Other (i.e.textiles)	2±1.3	24	20-40	3.81	3.71	3.87	3.79	3.83
Total	N/A	N/A	N/A	**156.59**	106.21	202.24	136.01	177.09

It should be noted that in 'garden waste' about 9% is considered to be wood (Trois and Jagath, 2011) and that under 'other' there are also wastes containing carbon (e.g. textiles, fine organic materials or materials that were co-mingled and could not be classified). For this study it was assumed that 10% of the waste classified as 'other' (or 2% of the overall composition) contains DOC in the form of textiles. This assumption was based on data collected for the City of Mafikeng (2011). Table 4.7 shows that on average about 156.6 kg of DOC are contained per tonne of wet waste in South Africa.

More detailed calculations for the carbon input for each municipality can be performed using the default carbon content of different waste components from IPCC (2006) and a higher degree of accuracy in calculations can be achieved at local level if the waste composition and the amounts of waste landfilled for that municipality are known. However, many of the South African smaller municipalities do not have weighbridges at landfill sites and do not sample the composition of the incoming waste. For these municipalities an 'average' South African composition will facilitate calculations of GHG at local level, however, the degree of accuracy of these calculations will be relatively low.

4.1.4.5 Carbon balances and GHG emissions for South African landfill sites – Calculation procedures and assumptions

It should be noted that a series of methodological approaches (Smith et al., 2001; U.S. EPA 2006 and IPCC, 2006) deriving emission factors for landfills consider that the carbon exiting landfill sites in the form of NMVOC and the carbon dissolved in leachate are very small and can be neglected in calculations. In addition, no work to quantify the contribution of N_2O and halogens to total GHG emissions from landfills has been published to date. Manfredi et al. (2009) caution against neglecting DOC lost in leachate, which they have modelled in the range of 1 to 4%, and highlight that carbon stored in a landfill site within the first 100 years should be also included in the carbon balance of a landfill site. Therefore, this study includes carbon stored and accounts for carbon lost in leachate, but does not include trace NMVOC, halogens and N_2O direct emissions from the degradation of waste due to lack of data. Based on the literature (Manfredi et al. (2009) and U.S. EPA (2006)) calculation parameters have been defined in order to perform carbon balances and are presented in Table 4.8.

Table 4-8: Calculation parameters for different South African landfill sites

Type of waste	DOC content (kg C tonne^{-1} ww)	Fraction of DOC dissimilated as LFG (D_{LFG})	Fraction of DOC dissimilated as leachate ($D_{Leachate}$)	Fraction of DOC stored in the landfill
"Average" South African waste	156.6	50%	4% for dumps 2% for other landfills	46% for dumps 48% for other landfills

Based on the work done by Manfredi et al. (2009) it was assumed that the biogenic carbon lost in leachate was 2% for engineered landfill sites due to the lining limiting wash-out, and 4 % for dumps due to the lack of lining. It was also assumed that the biogenic carbon washed out with leachate will oxidise in time to CO_2 and is neutral with regard to global warming.

Calculations of direct GHG emissions from dumps

An emission factor which includes carbon storage has been calculated for dumps in South Africa by following the methodology presented in detail by Manfredi et al. (2009) and following the same assumption that, on a mass basis, 55 % of the DOC which gives rise to LFG becomes CH_4 and 45% becomes CO_2. It was calculated by the following equation (1) adapted from Manfredi et al. (2009):

$$CH_4 \text{ generated}_{\text{with storage}} \text{ (kg)} = DOC * D_{LFG} * 55/100 * 16/12 \quad (1)$$

where 16/12 is the stoichiometric factor to convert carbon into CH_4.

In a similar manner the amount of biogenic CO_2 generated (including that escaping through leachate) can be calculated by using equation (2).

$$CO_{2\ \text{with storage}} \text{ (kg)} = \{(DOC * D_{LFG} * 45/100) + (DOC*D_{Leachate})\} * 44/12 \quad (2)$$

where 44/12 is the stoichiometric factor to convert carbon into CO_2.

To estimate GHG emissions from unmanaged dumps relative to controlled landfill facilities, IPCC (2006) and UNFCCC (2008) recommended the use of a methane correction factor (MCF) to account for the fact that a larger fraction of the waste is likely to decompose aerobically and thus not produce methane. Recommended MCFs are as follows:

- 0.8 for unmanaged solid waste disposal sites – deep and/or with high water table. This comprises all solid waste disposal sites (SWDS) not meeting the criteria of managed SWDS and which have depths of greater than or equal to 5 m and/or high water table at near ground level. The latter situation corresponds to filling inland water, such as pond, river or wetland, by waste.
- 0.4 for unmanaged-shallow solid waste disposal sites. This comprises all solid waste disposal sites not meeting the criteria of managed SWDS and which have depths of less than 5 m.

Although the authors have observed that shallow dumps are common in rural communities, reliable information regarding the characteristics of dumps in South Africa is scarce. Therefore, a worst case scenario was assumed (deep dumps and/or high water table) and a methane correction factor of 0.8 was used in this study.

Based on the assumptions presented above, the equation to calculate CH_4 emissions for South African dumps was defined as follows:

$$CH_4 \text{ emitted (kg)} = CH_4 \text{ generated} * MCF \quad (3)$$

Equation (1) was used to calculate the generated CH_4. To convert the amount of CH_4 emitted into CO_2 e, a global warming potential (GWP) of 25 (1 kg CH_4 = 25 kg CO_2) was used (IPCC, 2007).

Calculations for a second scenario were performed, in which carbon storage was assumed to be zero. In this case following equation was used:

$$CH_4 \text{ generated}_{\text{ no storage}} \text{ (kg)} = DOC * (1 - D_{Leachate}) * 55/100 * 16/12 \quad (4)$$

The CH_4 emitted for this second scenario was calculated using the CH_4 generated as per equation (4) and the MCF as per equation (3).

Calculation of direct GHG emissions from landfills without gas collection

The majority of landfills in South Africa do not have gas collection. For the calculation of direct emissions from these landfill sites, similar formulae were used as the ones presented in equations (1), (2), (3) and (4) for dumps. The differences are that in the case of landfill sites the MCF is 1, the carbon lost in leachate ($D_{Leachate}$) is 2% and a methane oxidation factor (MOF) has to be considered in order to account for the attenuation due to the cover of landfill sites.

Methane is oxidised by methanotrophic bacteria in the aerobic outer portion of the landfill cover material into methanol, which is further degraded into CO_2 (Spokas et al., 2006). Several researchers (Spokas et al., 2006; Chanton and Powelson, 2008 and Scheutz et al., 2008) show that significant portions (10-100%) of the methane present in the cover material can be oxidised. A few international studies (Smith et al., 2001; U.S. EPA, 2006 and IPCC, 2006) assume an oxidation rate of 10%. However, several researchers (Chanton and Powelson, 2008; Scheutz et al., 2008 and Manfredi et al., 2009) argue for the use of much higher oxidation rates. In particular, Chanton and Powelson (2008) show that in reality the oxidation rate of 10% is rather the exception and much higher oxidation rates were achieved in many sites in various locations in the USA. However, since there were no similar investigations conducted for South African conditions, in this study a conservative MOF of 0.1 (or 10% oxidation rate) was used.

Based on the assumptions presented above, the equation to calculate CH_4 emissions for South African landfills without gas collection was as follows:

$$\text{Emitted } CH_4 \text{ (kg)} = \text{Generated } CH_4 * \text{MCF}*(1\text{-MOF}) \qquad (5)$$

For the calculations taking into account carbon storage, equation (1) was used to determine the generated CH_4. For the calculations ignoring carbon storage, equation (4) was used to calculate generated CH_4.

Calculation of direct GHG emissions from landfills with gas collection and flaring

Only a few of the landfill sites in the large cities of South Africa are collecting and flaring LFG. In calculating an emission factors for these landfill sites, equation (1) and (4) was used to calculate CH_4 generated. In addition to this, important factors to be included in calculations are the collection efficiency for methane and the efficiency of the flaring system. Smith et al. (2001) consider a LFG collection efficiency of 54% for European landfills and U.S. EPA (2006) used 75%. Manfredi et al. (2009) use an overall average collection efficiency over 100 years which is 50-80% including post-closure emissions. These are instantaneous collection efficiencies and in international literature they show high variability, ranging from the 40s to 90s %, in the different studies in which they were measured (Spokas et al., 2006 and Barlaz et al., 2009). The two most important factors responsible for this variability are the type of landfill cover (i.e. daily, intermediate and final) used and the type of LFG collection system (i.e. horizontal and/or vertical) employed. Therefore, Barlaz et al. (2009) argue for a time-weighted average collection efficiency as opposed to an instantaneous collection efficiency. However, to calculate such a time-weighted average requires data that in many cases is not available in the current South African context. For example, for the Bisasar Road landfill site, the largest landfill in Africa and operating since 1985, the historical amounts of waste placed in early cells have not been recorded. LFG from these cells is collected through vertical wells. Therefore, for the current study an instantaneous collection efficiency of 75% was assumed. This was based on data from the first South African CDM project in the eThekwini Municipality (including the Bisasar Road and Mariannhill landfill sites) which showed that it is achievable in the local context (Moodley et al., 2010 and Moodley, 2012).

For the 25% CH_4 not collected and assumed to escape to the atmosphere, a MOF of 0.1 was applied. For the 75% CH_4 which is collected and flared, the efficiency of flaring was taken into account. Efficiency of flaring is considered 90% by Smith et al. (2001), 100% by USA EPA (2006) and 95-99% by Manfredi et al. (2009). In this study the efficiency of flaring was assumed to be 95%. This assumption was based on a worst case scenario. Data from the local CDM projects show much higher flaring efficiencies (98-99.9%) for the enclosed flares used. However, in the actual CDM accounting, these high efficiencies can be used only if continuous emissions monitoring equipment (gas analysers used to continuously test the flare emissions at the top of the flare shroud) is present. Without such monitoring equipment, the maximum efficiency allowable is 90%, even though it is known that the actual efficiency is 98-99% (Jewaskiewitz, 2012).

Based on the assumptions presented above, the equations to calculate CH_4 emissions for South African landfill sites with gas collection and flaring were as follow:

Emitted CH_4 (kg) = fugitive CH_4 from landfill site surface + CH_4 from flares (6)

Fugitive CH_4 from landfill site surface = Generated CH_4 * MCF*(1-α) * (1-MOF) (7)

where α = LFG collection efficiency factor and MCF=1.

CH_4 from flares = Generated CH_4 * α * (1-β) (8)

where β = flaring efficiency factor.

For the calculations taking into account carbon storage, equation (1) was used to determine the generated CH_4. For the calculations ignoring carbon storage, equation (4) was used to calculate generated CH_4.

Calculation of direct GHG emissions from landfills with gas collection and energy (electricity) production

Currently (2012) there are only three landfill sites in the country which collect LFG and use it for electricity production as part of a CDM project. These are the Mariannhill and the Bisasar Road landfill sites within the eThekwini Municipality (core city Durban) and the Chloorkop private landfill site in the greater Johannesburg area. Equations (1) and (4) were used to calculate the CH_4 generated with and without carbon storage. A collection efficiency of 75% was used. For the collected gas, the gas energy recovery efficiency was considered. This efficiency is assumed as 30% in this study, based on reported values of 25-35% by Manfredi et al. (2009) and 30% by Smith et al. (2001).

Avoided emissions are due to the replacement of electricity from the grid. Most of the electricity (around 90%) is generated in South Africa from coal. Therefore, local electricity provision (EP) carries a relatively high GHG burden (1.03 kg CO_2 e kWh^{-1}) as compared to the literature (Eskom, 2011). As a result the avoided emissions are higher per volume of CH_4 used. For the calculations it was assumed that the energy content (EC) of CH_4 is 37 MJ m^{-3} (assumptions also made by Manfredi et al., 2009), which is equal to 14.42 kWh kg^{-1} CH_4.

Based on the assumptions presented above, the equations used to calculate CH_4 emissions for South African landfill sites with gas collection and electricity production were as follows:

Emitted CH_4 (kg) = fugitive CH_4 from landfill site surface (9)

Fugitive CH_4 from landfill site surface = Generated CH_4 * MCF*(1-α)*(1-MOF) (10)

where α = LFG collection efficiency factor and MCF = 1.

In this case it was assumed that all of the collected CH_4 is combusted completely in the electricity generators and biogenic CO_2 results from this process. Avoided electricity GHG emissions for these landfill sites are calculated as follows:

Avoided emissions from electricity (kg CO_2 e) = - Generated CH_4 (kg) * α * EC * μ * EP (11)

where μ = energy recovery efficiency of generators

EC = energy content of methane (14.42 kWh kg^{-1})

EP = electricity provision factor (1.03 kg CO_2 e kWh^{-1})

For the calculations taking into account carbon storage, equation (1) was used to determine the generated CH_4. For the calculations ignoring carbon storage, equation (4) was used to calculate generated CH_4.

4.1.5. Results and discussions

The results and the related discussions are presented in 3 parts, namely the GHG emission factors derived for the collection and transport of municipal waste, direct GHG emission factors for landfills (including dumps) and indirect GHG emission factors for landfills.

4.1.5.1 GHG emission factor for the collection and transportation of municipal waste in South Africa

In order to see how South African municipalities compare in terms of diesel consumption and the resultant emissions, data was collected for four different municipalities and a summary of the results obtained is presented in Table 9. The average was weighted according to the mass of waste collected and transported to landfill sites.

Table 9 shows that the average amount of diesel consumed per tonne of wet waste collected and transported in South Africa is about 5 dm^3 (litres). This amounts to an emission of 14.6 kg CO_2 e $tonne^{-1}$ of wet waste. It should be noted that the City of Cape Town is the only municipality in the country which uses trains for the transport of some of its municipal waste.

A comparison between international diesel consumption data (see Table 4) and South African diesel consumption data (see Table 9) and resultant GHG emissions shows that, on average, South African data is comparable to the data collected from Taiwan. When compared with results from developed countries (e.g. Denmark and Canada) the diesel consumption and the resultant emissions are at the higher end of their ranges and similar to emissions from lower density urban areas. However, the Danish and the Canadian studies (see Table 4) did not include the entire transport distances and focussed mainly on collection. Unfortunately, these international studies did not publish enough details on to their study areas to allow a more in-depth, quantitative comparison.

Table 4-9: Diesel consumption and resultant emissions for waste collection and transport for selected South African municipalities

Municipality and characteristics	**Percentage distribution of households by type of refuse disposal***	**Diesel consumption ($dm^3 t^{-1}$ wet waste)**	**GHG emissions (kg CO_2 e t^{-1} wet waste)**	**Data quality**
eThekwini Municipality (core city Durban) Population: 3.6 mil Area: 2300 km^2 Waste collected: 1.65 mil tonnes	88.5 % collected and disposed by municipality 8.7 % communal or own refuse dumps 2.5 % no rubbish disposal 0.3 % other	4.2 (collection and transport by truck to landfill sites)	11.34	Full set for the last 3 years were available for amounts of waste and diesel used.
City of Cape Town Population: 3.5 mil Area: 2455 km^2 Waste collected: 1.66 mil tonnes	95.2 % collected and disposed by municipality 3.7 % communal or own refuse dumps 1.0 % no rubbish disposal 0.2 % other	5.5 (collection and transport by truck** to landfill sites)	14.85 - from collection and transport by truck 1.24 - from transport by electric train giving a **total of 16.09**	Full set for the last 3 years available for amounts of waste and diesel used. Emissions from transport by train included.
uMsunduzi Municipality (core city Pietermaritzburg) Population: 616 000 Area: 649 km^2 Waste collected: 169000 tonnes	72.4 % collected and disposed by municipality 24.6 % communal or own refuse dumps 2.7 % no rubbish disposal 0.2 % other	3.8 (collection and transport by truck)	10.26	Some data were missing in the last 3 years. Calculated monthly average values were used to fill gaps for amounts of waste missing.
Hibiscus Coast Municipality*** (core city Port Shepstone) Population: 225 000 Area: 839 km^2 Waste collected: 37400 tonnes	35.1 % collected and disposed by municipality 54.5 % communal or own refuse dumps 10.2 % no rubbish disposal 0.2 % other	5.6 (collection and transport by truck)	15.12	Only data for one year (2011) provided.
Average	N/A	4.9± 0.8	14.6±1.9	N/A

*This information was obtained from the Community Survey Report (StatsSA, 2007). The waste collected is disposed by municipalities in landfill sites engineered to different degrees.

** Electrical trains are used to transport waste for a distance of 37 km and an emission factor from the literature (0.056 kg CO_2 e per tonne per km – obtained from Eisted et al. (2009) - has been used)

*** This municipality contains a string of coastal towns and a rural component, which includes substantial tracts of productive commercial farmland and 6 tribal authority areas

In the local context, the difference in diesel consumption per unit of wet waste collected and transported is not that large between the two metropolitan municipalities and the two smaller municipalities considered, although the smaller municipality contained a large rural component and was expected to have much higher diesel consumption. This has to be viewed in connection to the low waste collection rates in rural municipalities in South Africa. In the Hibiscus Coast Municipality only 35.1 % of households have access to waste collection and in the the Ugu district (to which the Hibiscus Coast Municipality belongs) the overall access of households to refuse removal is 19.50 %, one of the lowest in the province (Municipal Demarcation Board, 2010). Waste is collected only from the households in the urbanised core of this municipality, with the majority of households in the tribal authority areas and on farmlands having no waste collection services at all. This waste is disposed in communal or private dumps which are not recorded or controlled. Therefore, waste collection vehicles do not drive to these areas, as in developed countries, and the waste generated by these households is not disposed in municipal permitted landfill sites. The reduced collection rate explains why this municipality has a relatively low consumption of diesel per unit of wet waste. Since most small, rural municipalities in South Africa have some of the lowest collection rates in the country, the trend observed in the literature (i.e. consumption of diesel increases in rural settings) seems not to be valid locally, because the majority of waste generated in these municipalities is not collected. Therefore, when considering emissions for the collection and transport of waste and using generation models (i.e. where quantities of waste are calculated based on a per capita rate) in South Africa, the collection percentage/frequency has to be considered. A much more reliable calculation, however, will be realised by using the amounts of waste which are reaching the landfills transported by municipal vehicles and the amounts of diesel used for collection and transport, as was done in the calculations in Table 4.9.

4.1.5.2 Direct GHG emission factors for landfill sites and dumps

Landfills have direct GHG emissions due to degradation of waste and operation of the landfill site and indirect emissions (i.e. in the construction of the site and after closure of the landfill, including rehabilitation of the area). Carbon emission relevant to South African landfill sites were calculated based on the methodology presented in Section 4.1.4.5 and are summarised in Table 4.10. However, biogenic CO_2 emissions resulting from the degradation of waste is considered part of the short carbon cycle and, therefore, usually not included in accounting GHGs. In this study it is included as an information item only.

Table 4-10: Carbon stored, CH4 and biogenic CO2 emissions per tonne of wet waste for the different types of landfills used in South Africa (kg tonne-1 of wet waste)

Type of landfill site	Carbon stored*	CH_4 emissions		Biogenic CO_2 emissions (including from leachate DOC)	
		With carbon storage	Without carbon storage	With carbon storage	Without carbon storage
Dumps	72.04	45.94	88.20	183.74	331.66
Landfill sites without gas collection	75.17	51.68	101.29	156.47	295.66
Landfill sites with gas collection and flaring	75.17	15.07	29.54	257.13	492.96
Landfill sites with gas collection and electricity generation	75.17	12.92	25.32	263.06	504.56

*Used only when carbon storage was taken into account.

This table presents emissions considering two different approaches. The first one neglects that over a 100 year period there is considerable carbon storage in landfill sites and the second approach takes this storage into account. The storage of biogenic carbon in landfill sites brings environmental benefits and it is argued (Manfredi et al., 2009 and Christensen et al. 2009) that it should become routinely included for waste management. To acknowledge that this carbon is sequestrated some authors (e.g. Manfredi et al., 2009) subtract it in the GHG emission calculations for landfill sites. However, other carbon accounting approaches for the waste sector do not include carbon stored in landfills at all. For example, the IPCC (2006) approach for inventories includes stored carbon in landfill sites only as an "*information item*" for the waste sector, the accounting includes it in the "Agriculture, Forestry and Other Land Uses" sector (IPCC, 2006).

Table 4.11 presents the overall direct GHG emissions. These were calculated by taking into account the CH_4 emitted (with a GWP of 25) and by subtracting the carbon sequestrated and the emissions from the avoided electricity due to LFG utlilisation. Carbon storage is neutral if the carbon is just sequestrated (i.e. waste does not decompose and does not generate methane). However, the stored carbon is taken out from the carbon cycle and, therefore, can be subtracted in the overall carbon balance. Both scenarios are presented.

Table 4-11: Overall direct GHG emissions from South African landfill sites (kg CO2 e tonne^{-1} wet waste)

Type of landfill site	Emissions avoided due to sequestrated carbon	Emissions avoided due to electricity generation	CH_4 emissions		Overall direct GHG emissions with neutral carbon storage		Overall direct GHG emissions with carbon storage subtracted	
			With carbon storage	Without carbon storage	With carbon storage	Without carbon storage	With carbon storage	Without carbon storage
Dumps	-264.13	0	1148.40	2204.93	1148.40	2204.93	884.27	2204.93
Landfill sites without gas collection	-275.62	0	1291.95	2532.22	1291.95	2532.22	1016.33	2532.22
Landfill sites with gas collection and flaring	-275.62	0	376.82	738.56	376.82	738.56	101.20	738.56
Landfill sites with gas collection and electricity generation	-275.62	-191.89	322.99	633.06	131.10	441.17	-144.52	441.17

The factors calculated, taking into account carbon storage, are in the range published in the literature (Barton et al, 2008; Manfredi et al. 2009) and differences are due to the biogenic carbon content of the waste. The subtraction of carbon storage makes a substantial difference for all landfill scenarios and, therefore, it is important for any GHG accounting exercise to report if storage was included and how.

As shown in Table 4.11, landfill sites without gas collection have the highest emissions and this agrees with previously published studies (see Friedrich and Trois, 2011 for a summary). These landfill sites need LFG collection system with flaring or electricity generation in order to reduce their GHG emissions. However, the construction of such systems involves financial investments and technical capacity that are not available in the majority of South African municipalities. All LFG collection and utilisation systems are dependent on the carbon credits sold and/or on subsidies for renewable energy accessible through the South African Department of Energy. The first LFG to electricity initiative in South Africa, in the eThekwini Municipality, is a CDM project and depends on the financial support from the World Bank's

Prototype Carbon Fund (Couth et al., 2011). The only other LFG to electricity project in the country (a private landfill site in Johannesburg) is also a CDM project. However, extending this kind of initiatives to other landfill sites in the country is problematic and municipalities are faced with financial uncertainties. Firstly, the international agreements on how to replace CDM arrangements after 2012 when the Kyoto Protocol ends are still under negotiation. Secondly, even if the uncertainties related to carbon trading are resolved, there are technical issues linked to the size of landfill sites and the amount of LFG generated. Couth et al. (2011) have shown that small to medium sized CDM landfill gas utilisation projects are not viable in Africa without a renewable energy tariff. Such tariffs have been introduced in South Africa in 2009 and in 2011 they have been lowered. Resolving these financial uncertainties at international and national level would help many municipalities to better plan for such projects and could lead to significant GHG savings.

The low waste collection rates in the rural fringes of South African municipalities have important consequences for the disposal of uncollected waste in dumps and the associated degradation process and GHG emissions. Dumps have important negative impacts on the surrounding areas (e.g. surface and groundwater pollution, air pollution, odours, etc.) and they present health risks to the communities nearby. Therefore, municipalities are extending access of households to waste collection services and the use of controlled landfills for disposal. However, without LFG collection and utilisation on landfill sites this positive development will cause higher GHG emissions. In South Africa the GHG emissions from these dumps have been so far ignored. In inventories performed at national level (GHG Inventory, 2010) and at local level (e.g. eThekwini GHG Inventory, 2011) they were not included and two reasons were given. "*Firstly, data on waste dumped in unmanaged and uncategorised disposal sites have not been documented. Secondly, most of the unmanaged and uncategorised disposal sites are scattered throughout rural and semi-urban areas across South Africa and are generally shallow (i.e. less than 5 m in depth). In such shallow sites a large fraction of the organic waste decomposes aerobically which means methane emissions are insignificant compared to those from managed landfill sites*" (GHG Inventory, 2010, pp. 68). Such an approach is superficial and, therefore, there is an important need to investigate dumps and associated GHG emissions. The emission factor developed in this study for dumps enables some estimation of GHG emissions. However, it should be seen as a starting point and the factor should be refined and improved once more data is collected.

A simple sensitivity analysis was conducted to test the variability of the results to the calculation inputs, and the calculations procedures were run separately with changed DOC (maximum and minimum), increased MOF and increased and decreased collection efficiency. An increased DOC lead to less GHG emissions from landfill sites with gas collection systems. It had the opposite effect for landfills without such systems. Changing the DOC and the collection efficiency changed the magnitude of the outputs but did not change the order of GHG emissions for the different landfill types. Landfill sites without gas collection were having the highest GHG emissions with maximum and minimum DOC, followed by dumps, landfill sites with flaring and landfill sites with electricity generation. However, increasing the MOF from 0.1 to 0.4 showed that all landfill sites are having less GHG emissions than dumps. This is an important observation, in particular for landfill sites without LFG collection systems, because even lacking financial resources for collection systems, GHG emissions can be substantially reduced using cover materials which enhance methane oxidation. More local research has to be conducted in this field.

4.1.5.3 Indirect GHG emission factors from landfill sites and dumps

Data used to calculate the upstream, indirect emissions have been sourced from the literature and locally, from interviews with waste managers from the eThekwini and uMsunduzi Municipalities and are presented in Table 4.12. The quantities of sand, gravel and crushed rock used locally are higher than the quantities cited in the literature, but the amounts of electricity and diesel used for daily operations of landfill sites are lower. However, because these parameters have been investigated only for three local landfill sites, higher values from literature (2 kWh and 1 dm^3 diesel per tonne of wet waste) were used in calculations. Also, the one local landfill site used geosynthetic clay liner, which has much lower GHG emissions associated (0.22 kg CO_2 e kg^{-1} liner giving 0.51 kg CO_2 e $tonne^{-1}$ wet waste) as compared to a high density polyethylene (HDPE) liner used in the literature. By using these parameters and the associated emission factors from Manfredi et al. (2009) the indirect emissions from landfill sites have been calculated as 9.53 kg CO_2 e per tonne of wet waste.

Table 4-12: Indirect GHG emissions for South African landfill sites

GHG emission type	Amount used per tonne of wet waste from literature (Manfredi et al., 2009)	Amount used per tonne of wet waste for a local landfill site	kg CO_2 e per tonne of wet waste for local calculations	Observations
Diesel for construction of cells on landfill site	0.5-1 L	No records for South Africa	1L*2.7 = 2.7	Higher value (worst case scenario) from literature used.
Synthetic liner for construction of cells HDPE (2mm thick)	1 kg (assuming a cell depth of 20 m)	No records for South Africa	1kg*1.9 = 1.9	
Provision of sand, gravel or crushed rock needed on site	0.1 tonne	0.4 tonne	0.4 t*0.5 = 0.2	Local value was obtained as an average over 3 years from 3 eThekwini Municipality landfill sites
Electricity for on-site lighting, administration buildings, pumps and fans	2-12 kWh (Niskanen et al., 2009 in Manfredi et al., 2009)	0.9 kWh	2 kWh*1.015 = 2.03	Local value was obtained as an average over one year from one eThekwini Municipality landfill site with leachate and gas collection systems. Local electricity factor used.
Diesel for daily on-site operations	1-3 L	0.34 L	1 L* 2.7 = 2.7	Local value obtained from one landfill site over one year.

The indirect upstream emissions are minor compared to the direct emissions for most types of landfill sites and this result is similar to the findings of Manfredi et al. (2009) for European landfills.

The downstream indirect emissions from fossil fuels used on landfill sites have been considered to be zero, although when a landfill site is closed, diesel will be consumed for landscaping and energy will be used by monitoring equipment. However, the GHG resultant are considered very small when calculated per unit of waste placed in that landfill site and in this study are ignored. However, the downstream indirect emissions due to electricity generation from LFG have been included as presented in Table 4.11.

It should be noted that in terms of upstream GHG emissions, it is assumed that no excavation was undertaken in order to establish the dumps and no GHG emissions are assigned due to the construction of dumps. However, downstream indirect GHG emissions due to dumps are problematic, since some of these dumps have to be cleared (especially the illegal ones within metropolitan areas) due to their environmental impacts. There is no literature on GHG emissions of waste which partially decomposes in a dump (i.e. for a few years) and is then transferred to a landfill site. Therefore, this area needs more research, especially in the South African context where municipalities, faced with increased legal pressures (Waste Act of 2008 see RSA, 2008) strive to improve their waste management operations and dumps are less tolerated than in the past.

Indirect, upstream emissions due to landfill sites with gas collection and flaring should be slightly higher compared to those without gas collection, since energy and materials were used in the production of the components (pipes, pumps, fans, etc) of the collection system and in the construction of the system itself. However, so far there is no quantification of these elements in the literature and no records could be sourced from the few South African landfill sites. In fact the electricity consumption for one local landfill site with gas collection was much lower than the range presented in the literature (see Table 4.12). Thereafter, more research and data from more landfill sites are needed for such quantifications to be representative for the country. The energy (i.e. electricity) for operating the collection system will be included in the overall energy requirements of the site, and in the case of the eThekwini Municipality landfill sites investigated it represents a very small percentage.

4.1.6. Conclusions

The development of emission factors for waste management processes is important for municipalities in developing countries, because many of these municipalities do not have all the data needed for a thorough GHG accounting process. However, few developing countries have attempted developing emission factors to suit specific national conditions with the result that almost all published factors emanate from Europe and North America. The current research was undertaken to fill this gap in the context of South Africa, and a series of emission factors have been calculated for representative waste management processes in this country. This paper presents the development of emission factors for collection and transport of municipal waste as well as for several disposal alternatives.

For collection and transport of waste, the average diesel consumption is about 5 dm^3 (litres) per tonne of wet waste collected and transported in South Africa and the average GHG emissions are about 15 kg CO_2 e. These averages are in the range published in the international literature for lower density urban areas. However, for the collection and transport of municipal waste in the rural fringes of municipalities, South African GHG emissions are much lower than the international ones. This difference is due to the lower collection rate in these locations and not due to higher efficiencies in the collection processes. As collection services for municipal waste are expanded, the collection rate will increase and it is expected that higher overall GHG emissions will be calculated for this process.

The waste which is not collected ends up in uncontrolled dumps, which have important negative impacts for the surrounding communities and the environment. Therefore, in short term improving the quality of life and improving waste management in South Africa is expected to generate more GHG emissions. These can be reduced through LFG collection and utilisation. In the South African context, where significant amounts of waste are still disposed in dumps, more data and research is needed on the dumps themselves, as well as on the overall performance of landfill sites in the presence and in the absence of LFG collection systems. In particular, methane oxidation in landfill covers for landfills without gas collection systems needs investigation as it has the potential to significantly reduce GHG emissions from these sites without needing high investments.

The GHG emission factors developed for landfill sites show that the highest contributions per unit of wet waste in this country are due to landfill sites without gas collection. These are the majority of the landfill sites in the country and with the upgrading of dumps, more and more of the local municipal waste will be disposed in this way. This situation is valid for the majority of municipalities in the developing world and will lead to negative consequences for the emission of GHG. Therefore, more efforts should be made to reduce these emissions by designing covers to maximise methane oxidation and by collecting and utilising LFG. However, lack of financial resources is limiting the implementation of LFG collection systems in South Africa. Possible financial avenues to increase the implementation of such systems are the recently introduced renewable energy tariffs and the CDM projects. The CDM projects, which are instrumental for LFG collection and electricity production in the country proved very successful, and these landfill sites have the lowest GHG emissions. However, future extension of these projects to other landfill sites is uncertain. Currently there is no other mechanism to replace the Kyoto agreements and the subsequent CDMs, which provide an external source of funding for LFG collection. For cash-strapped municipalities in the developing world this is a negative outcome.

Another important conclusion from the development of GHG emission factors for landfill sites is that inclusion or exclusion of carbon storage is important for calculations. More studies should be undertaken in order to develop a common approach to future GHG emission calculations. In the meantime, GHG reporting from landfill sites should give enough details on the calculation procedures and how emission factors have been aggregated.

Acknowledgements

The input from one of the anonymous reviewers is greatly appreciated. The feedback received helped in improving the paper substantially. Our gratitude is also extended to Mr Chris

Brouckaert for his help with the Monte Carlo simulations and to Mr John Parkin and Prof Holger Friedrich for proof reading.

5.1.7. References

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4.2 GHG Emission Factors Developed for the Recycling and Composting of Municipal Waste in South African Municipalities

4.2.1. Abstract

GHG (greenhouse gas) emission factors for waste management are increasingly used, but such factors are very scarce for developing countries. This paper shows how such factors have been developed for the recycling of glass, metals (Al and Fe), plastics and paper from municipal solid waste, as well as for the composting of garden refuse in South Africa. The emission factors developed for the different recyclables in the country show savings varying from -290 kg CO_2 e (glass) to – 19111 kg CO_2 e (metals - Al) per tonne of recyclable. They also show that there is variability, with energy intensive materials like metals having higher GHG savings in South Africa as compared to other countries. This underlines the interrelation of the waste management system of a country/region with other systems, in particular with energy generation, which in South Africa, is heavily reliant on coal. This study also shows that composting of garden waste is a net GHG emitter, releasing 172 and 186 kg CO_2 e per tonne of wet garden waste for aerated dome composting and turned windrow composting, respectively. The paper concludes that these emission factors are facilitating GHG emissions modelling for waste management in South Africa and enabling local municipalities to identify best practice in this regard.

4.2.2. Introduction

South Africa is a developing country with a population of about 50.5 million people (StatsSA, 2012) and is seen as the economic powerhouse of the Southern African region with the highest gross domestic product (GDP) of the continent (Wikipedia, 2012). Its development relied heavily on historically cheap energy from fossil fuel, and coal combustion produces 87% of the electricity used in the country (Eskom, 2012). As a result, South Africa is ranked 13 in the global list of GHG (greenhouse gas) emitters being responsible for 1.45% of the global emission (UN – Millennium Goals, 2012), emitting about 437.3 million tonnes CO_2e in the year 2000 (GHG Inventory, 2010). In 1997 the country ratified the United Nation Framework Convention on Climate Change (UNFCCC) and as a result it accounts and reports for GHGs at national level. Furthermore, although not bound by the Kyoto Protocol, a series of initiatives at regional and municipal level have been undertaken for the accounting, reporting and reduction of GHGs, including voluntary and CDM (Clean Development Mechanisms) initiatives within the waste sector (Friedrich and Trois, 2010). In this context GHG accounting and associated emission factors are becoming increasingly important, however, for waste management such factors are lacking for South Africa and in general for developing countries. The aim of this study was to develop GHG emission factors for solid waste management in South Africa and the results are presented in two parts. We have previously reported (Friedrich and Trois, 2013) the GHG emission factors for collection, transport and landfilling of municipal waste and the current paper reports these factors for recycling and composting. Glass, metals (Al and Fe), plastics and paper were chosen, as these are the main recyclables. Together they represent about 31% of the 53 million tonnes general solid waste in the country (Republic of South Africa, Department of Environmental Affairs – RSA DEA, 2012) and about 41% of municipal solid waste when construction and demolition waste is excluded (Friedrich and Trois, 2013).

International studies (Smith et al., 2001; United States Environmental Protection Agency – US EPA, 2006; Merrild et al., 2009a; Astrup et al., 2009; Damgaard et al., 2009; Larsen et al., 2009b) show that recycling of different fractions of the municipal solid waste usually results in important GHG savings. However, the extent of the savings that can be achieved for a particular material, in a particular municipality depends heavily, not only on the waste management system itself, but also on other interlinked systems, most notably the energy system of that region/country. In particular, recycled materials which replace virgin materials needing large amounts of energy in the production process have the potential to bring about the largest GHG savings. This should be emphasised in a country like South Africa which depends largely on coal for energy generation and which intends to reduce its overall GHG emissions. Therefore, to enable locally relevant GHG calculations it is important to quantify these GHG savings due to recycling.

Composting is the aerobic degradation of the organic fraction of municipal waste and can realise savings but also net GHG emissions. This depends on a variety of parameters, which define the process and on the applications for the resultant compost (Boldrin et al. 2009). Unlike recycling which in terms of GHG emissions is linked to the energy system of a country, composting is more prominently linked to the agricultural system where it is used. The application of compost to land "imply a time limited sequestration of carbon that could inform policy on climate change while indirectly reducing GHG emissions by reducing demand for, and application of materials such as mineral fertilizers, pesticides and peat" (Favoino and Hogg, 2008). Also the use of compost can lead to many positive, indirect aspects related to increased potential of soils to retain moisture, reduced irrigation needs during droughts and improved workability of soils. These benefits which also save GHG have not yet been quantified. However, it has to be underlined that the organic fraction of the municipal solid waste, which in developing countries is much higher (Friedrich and Trois, 2011), still poses some challenges in terms of generating good quality compost. In South African municipalities, as in many other parts of Africa and in similar developing countries, the garden/yard waste stream is seldom separated from other municipal domestic waste. However, when separated collection occurs, it is generally limited to woody/leafy waste material from parks and households. This garden refuse, if aerobically composted, produces a stabilised product with lower C/N ratio (generally lower than 21) and higher pH, that can be used to neutralise acidic soils but does not display similar nutrient qualities as organic waste compost (Trois and Polster, 2007).

4.2.3. GHG emission factors for recycling and composting – An overall perspective of the approaches and methodologies followed

An emission factor is a term used to present the amount of GHGs released per unit of mass, volume or energy (Merrild et al., 2009a). They are also defined as "generic and highly aggregated factors determined from a number of processes, representing similar characteristics" (Gentil et al., 2009). GHG emission factors for recycling and composting have been developed for the Northern Hemisphere in the last 20 years. They have been derived by using life cycle assessments (LCAs) with different degrees of sophistication. They all take into account upstream, operating and downstream emissions, but there are differences in how boundaries are set. For example, some studies include transport in the upstream emissions (e.g. US EPA, 2006) and others separate transport from recycling and composting processes (e.g. Astrup et al., 2009; Boldrin et al. 2009; Damgaard et al., 2009; Larsen et al., 2009 and Merrild et al., 2009a). In this paper, collection and transport of recycled materials and organics to be composted were included in the calculation of the emission factors and are clearly presented in the relevant sections.

An exhaustive summary on the GHG emitted in the different life cycle stages of each waste management process is presented by Scheutz et al. (2009) and given in Table 4.13. It follows the LCA conceptual approach of Gentil et al. (2009), which was also followed in this study.

Table 4-13: Qualitative GHG emissions from recycling and composting of municipal solid waste (adapted from Scheutz et al., 2009).

Upstream indirect emissions	***Operating direct emissions***	***Downstream indirect emissions***
Sorting, pre-treatment and recycling		
Emission of CO_2, CH_4, N_2O due to: collection and transport of recyclables, production of fuel used in recycling facilities, heat and electricity consumption and infrastructure	CO_2, CH_4, N_2O, trace CO and NMVOC from fuel combustion	Substitution of raw materials and avoided GHG emissions from material production. Rejects (emission depends on the type of secondary technologies)
Composting		
Emission of CO_2, CH_4, N_2O due to: collection and transport of organic waste, production of fuel used in composting facilities, heat and electricity consumption and infrastructure	Emissions from windrows, reactors and biofilters (CO_2bio, CH_4, N_2O) CO_2, CH_4, N_2O, trace CO and NMVOC from fuel combustion	Carbon bound in soil (100 years) Substitution of growing media (NPK fertilizers, peat) Avoided N_2O and CO_2 from the reduced production of fertilizers and avoided emissions from peat utilisation.

Data on the waste management processes investigated in this paper (composting and recycling) were obtained from published sources, as well as through interviews, site visits and direct observations. Table 4.14 presents the types of data used and their sources, as well as some observations regarding data quality.

Table 4-14: Types of data used, sources of data and observations regarding quality of data used

Type of data	Source	Observations
Production of virgin materials to be replaced by recycled waste	Glass – Consol and Nampak Packaging Steel – South African Iron and Steel Institute and Arcelor Mittal SA Aluminium – Aluminium Federation of South Africa and the Carbon Disclosure Project Report (2011) Plastics – Plastics SA (previously Plastics Federation of South Africa) and Plastics Europe Paper – Paper Manufacturers Association, South Africa and some relevant companies (Mondi SA, Sappi SA one tissue paper manufacturer and three independent cardboard manufacturer)	Data on production processes were obtained locally, however, for very few speciality materials needed in the production process, international data sourced from the SimaPro7 and the Ecoinvent databases were used.
Generation and distribution of electricity	Eskom - South Africa's monopoly electricity generator and distributer	Country specific, South African electricity mix including losses in the transmission and distribution network.
Data on carbon content of garden and food waste used for composting	A series of local studies undertaken at the University of KwaZulu-Natal (UKZN)	Detailed data collected over 3 years were used.

Initially LCA modelling based on these data was performed with the help of the SimaPro 7.2 tool, which contained the SimaPro 7.2 and the Ecoinvent data bases. Because this tool uses highly aggregated international LCA data, additional calculations were performed, in which only disaggregated local GHG data was used. It was decided that the disaggregated method of calculation is preferred because of transparency. It was in line with the methodology used by the majority of the international LCA tools and studies (see Friedrich and Trois, 2013 for a summary). Individual assumptions and calculation procedures are briefly presented in each of the following sections.

4.2.4. The development of GHG emission factors for recycling

South Africa is a country with accentuated social disparities and this is also mirrored by the recycling activities in the country. Recycling is market driven. On the one side there is a sizable informal sector, which collects and separates recyclables out of the municipal waste and sells them to buy-back centres operated by municipalities or to individual entrepreneurs, which transport recyclables and in turn sell them to formal recycling companies. On the other side there are formal collection arrangements, most notably drop-off (or “bring”) containers and drop-off centres and kerbside collection in certain municipalities or parts thereof. There are only a few operational material recycling facilities (MRFs) in the country (e.g. in Cape Town) and a negligible proportion of the municipal solid waste is recycled via this route. A voluntary deposit-return system operates in some sectors, for example for some refillable bottles and for certain plastic crates, but there is no compulsory deposit-return system in the country (PACSA, 2011). There is a series of organisations which emerged from the industry (most notably The Glass Recycling Company, Collect-a-Can (metal), the Paper Recycling Association of South Africa (PRASA), Plastics SA and the Polystyrene Packaging Council) which have started to play a more active role in engaging with government (at policy level) and municipalities (in partnerships) in many recycling activities.

Recycling activities in South Africa are expected to increase, driven by a variety of factors of which the legislative pressures set by the Waste Act (Act 59 of 2008) needs to be mentioned. This Act places responsibility on the State to “put in place uniform measures that seek to reduce the amount of waste that is generated and, where waste is generated, to ensure that the waste is reused, recycled and in an environmentally sound manner before being safely treated and disposed of” (RSA - Waste Act, Act 59 of 2008). It stipulates that all levels of government (central, provincial and local) have responsibilities, as do the producers of waste and waste holders. As a result of this act, the Packaging Council of South Africa (PACSA) released a sectorial plan on how recycling rates for post-consumer waste should increase for each waste stream and the current, as well as the target figures, are presented in Table 4.15.

Table 4-15: Current and targeted recycling figures for South Africa (Source: PACSA, 2011)

	2009 Actual	Year 2014 Target
Paper	56%	61%
Metal	56%	65%
Glass	32%	43%
Plastic - PET	26%	37%
Plastic - Other	28%	35%
Overall recycling rate	44.5%	51%

To calculate the carbon expenditure and saving for each of the recyclable materials found in the municipal solid waste, a carbon balance by means of a simplified LCA was undertaken for each of these materials in the South African context. As such the energy input in the manufacturing of virgin materials (glass, paper, metals (Al and Fe) and plastics) was quantified and the source of that energy established. In the case of electricity, local emission factors for the production of electricity were used. Taking into account the most recent recycling rates for South Africa (PACSA, 2011), as well as general losses in the recycling process, replacement values (energy and the associated GHG) for virgin materials were calculated depending on the type of recycling (open loop vs. closed loop).

4.2.4.1 Recycling factor for glass

Glass represents about 7% of the municipal solid waste stream in South Africa (Friedrich and Trois, 2013) and is made up of packaging glass from food (e.g. jars) and beverages (e.g. bottles) as well as flat glass from demolition waste. These kinds of glass are made from soda-lime glass for which the main inputs are quartz (SiO_2), sodium oxide (Na_2O) and calcium oxide (CaO) (Larsen et al., 2009). In 2009/10 about 295879 tonnes were recycled annually in the country (PACSA, 2011). There are two glass packaging manufacturing companies in South Africa and they are the users of the recycled glass (cullet) in the country. These two companies, together with about 20 filler companies, (representing about 80% of all new glass containers purchased in the country) have established The Glass Recycling Company in 2005, which aimed to increase glass recycling. They have succeeded in increasing the recycling rate for glass from 18% in 2005/6 to 32% in 2009/10. This was achieved by placing more glass banks, by assisting 900 entrepreneurs (mostly from disadvantaged communities) to purchase glass from collectors and by running educational campaigns (PACSA, 2011). The funds for these activities, as well as for technological improvements (i.e. automated colour separators) which are needed to cope with increased volumes of glass recycled, have been raised by voluntary levies paid by the fillers and the glass manufacturers themselves.

Taking into account that for the year 2009 about 1 000 000 tonnes packaging glass was produced in South Africa, and that 25 000 tonnes were imported and 125 000 tonnes exported, the local consumption of packaging glass was 905 000 tonnes. Of the glass produced about 40% was flint (clear glass), 20% amber and 40% the five different shades of green (PACSA, 2011). There are some logistical colour imbalances in the recycling process which impose limitations; with most of the green glass being manufactured in the wine region of South Africa but consumed elsewhere. In and around Johannesburg, for example, are only limited opportunities for recycled green glass, but a lot of it available. Transport back to the wine regions, which are around Cape Town, is around 1400 km and at the moment it is not commercially and environmentally viable (PACSA, 2011).

Currently (2012) all packaging glass recycled is used by the two glass manufacturers. Glass is not colour sorted at household level, and both glass manufacturing companies have invested in recycling operations with one company building a new state of the art recycling plant in 2010 in the Johannesburg area (which breaks, cleans and colour sorts glass automatically) and the other company investing in expanding existing operations. Theoretically, glass can be recycled indefinitely with little loss in the process; however, in practice there are serious limitations due to the logistics of collection and the colour separation process which is undertaken at the glass manufacturer level. Even the best performing cullet automatic sorting plants will generate about 43% of unidentifiable mixed cullet which can be processed only into green glass which, as recycling increases, probably will exceed the market demand for such glass (PACSA, 2011). Based on the information presented above, it was assumed that post-consumer glass recycling in South Africa is a closed loop process. It was also assumed that a similar process is used for the manufacture and recycling of flat (plate or float) glass which ends up in the municipal solid waste. The recycling factor calculated for recycled glass is presented in Table 4.16.

Table 4-16: The calculation of a GHG emission factor for glass recycling in South Africa

Emission factor calculation	**Source, reference or conversion**
Indirect, upstream processes 1) Collection of glass (1,070 t if 7% reject considered) Assuming 5-10 km one way carrying 100 kg (as done by local entrepreneurs) by personal car or small van 1,070 t*45 kg CO_2 e /tonne = 48.2kg CO_2 e 2) Transport of glass Assuming 200 km in a truck (≥ 16 tonnes) 1,070t *200km*0.190 kg CO_2e / tonne / km = 40.7 kg CO2 e ***Total 88.9 kg CO_2 e***	Reject of incoming glass obtained from Larsen et al. (2009b). Car emission factor obtained from Eisted et al. (2009). Average distance obtained from a local glass recycling company. Truck emission factor obtained from Spielmann et al. (2004) cited in Eisted et al. (2009). Worst case scenario taken since South Africa has lower emission regulations for trucks (e.g. EURO 0, I and II, as opposed to EURO III, IV and V)
Direct, recycling processes Sorting and cleaning of glass and the production of cullet 1,070 t * 22 kWh / tonne = 23.5 kWh 23.5 kWh* 1.03 kg CO2e /kWh = ***24.2 kg CO_2e***	Energy needed for the process obtained from a local glass manufacturer. Average emission for South African electricity generation and transmission obtained from Eskom (2011) - the local monopoly energy producer.
Indirect, downstream processes 1) Energy savings due to lower melting point (i.e. the endothermic reactions of converting carbonates into oxides do not happen when using cullet) On average 1.57 GJ are saved in the form of oil, gas and electricity. This amounts to 164 kg CO_2 e saved per tonne of cullet used. 2) Avoided calcination emissions 180 kg CO_2 e saved per tonne of cullet used 3) Avoided emissions due to the provision of raw materials (1.2 t feedstock consisting of 240 kg soda ash, 240 kg limestone and 720 kg sand) 60 kg CO_2 e saved **Total savings = 404 kg CO_2**	Data from a local glass manufacturer. Data obtained from a local glass manufacturer. Data obtained from a local glass manufacturer; however, additional modelling of the transport of some of these materials had to be undertaken.
Overall total 88.9+24.2-404= ***- 290.1 kg CO_2e***	

4.2.4.2 Recycling factors for metals (Al and Fe)

In South Africa, metals constitute about 4 % of the municipal waste stream (Friedrich and Trois, 2013) and are mainly of post-consumer origin. Ferrous metals (i.e. iron and steel) and non-ferrous metals (i.e. aluminium, lead, copper, zinc, precious metals) contained in many post-consumer objects will end up in the municipal waste, however, steel and aluminium are dominating and emission factors have been calculate only for these two metals.

Aluminium (Al) in the Southern African region is produced in 3 separate smelters belonging to the international company BHP Billiton. Two of these independent operations are situated in Richards Bay, South Africa and about 600 000 of the 815 000 tonnes primary aluminium produced by these two operations are exported and sold on global markets. The remainder of 215 000 tonnes primary aluminium is used by the South African industry and an additional 135 000 tonnes of aluminium components (finished products and materials used) are imported. In turn 179 000 tonnes aluminium products are exported. Therefore, the South African market in 2011 used about 235 000 tonnes aluminium that is theoretically recyclable, of which in that year 66 000 tonnes was collected and recovered (average age of recycled Al is 12 years) giving a recycling rate of about 28% for this metal. From the recovered aluminium 41 000 tonnes were exported as scrap and 3 000 tonnes as secondary ingots (Aluminium Federation of South Africa, 2012). There is no data on how much aluminium is used for packaging and how much of that is recycled. Aluminium fetches high prices at scrap dealers and this is the reason why many post-consumer aluminium objects (e.g. furniture, frames, cans etc.) are recycled, however, this is not valid for all post-consumer aluminium waste (e.g. foil products) which then ends up in landfill sites. Overall, aluminium is considered to represent only a small proportion of total metal packaging in South Africa (PACSA, 2011). The recycling factor calculated for aluminium is presented in Table 4.17.

South Africa has five primary carbon steel producers (of which ArcelorMittal is the largest) and one primary stainless steel producer (South African Iron and Steel Institute, 2012). Steel used for packaging represented about 5% (256 744 tonnes of the total 5 336 888 tonnes) of the total steel sales in the country in 2010 (South African Iron and Steel Institute, 2012).

Table 4-17: The calculation of a GHG emission factor for the recycling of aluminium in South Africa

Emission factor calculation	**Source, reference or conversion**
Indirect, upstream processes 1) Collection of aluminium (1 t of scrap with a rejection rate of 5% at sorting stage) Assuming 5-10 km one way carrying 100 kg (as done by local entrepreneurs) by personal car or small van 1 t*45 kg CO2 e /tonne = 45 kg CO_2 e 2) Transport of aluminium Assuming 700 km in a truck (≥ 16 tonnes) 1 t *700km*0.190 kg CO2e / tonne / km = 133 kg CO_2 e ***Total 178 kg CO_2 e***	Reject of aluminium scrap obtained from Damgaard et al. (2009) Car emission factor obtained from Eisted et al. (2009). Truck emission factor obtained from Spielmann et al. (2004) cited in Eisted et al. (2009). Worst case scenario taken since South Africa has lower emission regulations for trucks (e.g. EURO 0, I and II, as opposed to EURO III, IV and V)
Direct, recycling processes Sorting 1 t*6.8 L diesel / tonne = 6.8 L diesel used by cranes 6.8 L* 2,7 kg CO_2 e / L diesel= 18.4 kg CO_2 e Shredding 0.95t*50kWh/tonne=47.5 kWh 47.5* 1.03 kg CO_2 e /kWh= 48.9 kg CO_2 e ***Total 67.3 kg CO_2 e***	Damgaard et al. (2009) Damgaard et al. (2009)
Indirect, downstream processes Savings from avoided production of aluminium 0.95 * 20375 kg CO_2 e/tonne = 19356 kg CO_2 e ***Total savings 19356 kg CO_2 e***	Data on GHG emissions per tonne of aluminium were calculated by using the emission data published in the Carbon Disclosure Project JSE Report (2011) by BHP Billiton.
Overall total 178+67.3-19356= **-19110.7 kg CO_2 e**	

The largest producer of steel packaging for food and beverages in the country is Bevcan Nampak and the South African market is dominated by steel cans. For example, the beverage cans manufactured and distributed in the country are made of 93% steel (the body) and 7% aluminium (the top cover end). On the recycling market they fetch a lower price compared to higher content aluminium cans which enter the country through imports. The major player in post-consumer metal recycling is Collect-a-Can, which is a joint venture company formed by ArcelorMittal South Africa (the largest steel producer of the country and the continent) and Nampak (the largest packaging company and beverage can manufacturer of the country and the continent). It was established in 1993 and has grown from an operation based around Johannesburg into a country-wide collection and recycling system, operating also in Botswana and Namibia. Initially it targeted beverage cans but later expanded to collect all types of metal cans. More recently independent scrap dealers are also becoming involved with steel recycling and pay for collected cans. The recycling rates of steel beverage cans reported has fluctuated, increasing from 18% in 1993 to 72% in 2008, however there was a drop due to the worldwide economic recession in 2009 to 69%, but a recovery to 70% in 2010 was observed. Overall metal recycling from post-consumer waste represented about 147 000 tonnes in 2009 (55.8 % recycling rate) and increased to 148282 tonnes (59.9 % recycling rate) during the year 2011 (PACSA, 2011 and Marthinusen, 2013). A decline of 1.2 % per annum in tonnage for metal packaging has been forecasted and is currently (2012) observed, so as recycling rates will increase only a modest increase in tonnage of recycled metals was recorded and is expected in the next few years (PACSA, 2011 and Marthinusen, 2013).

The steel cans collected from municipal waste in the country are transported to a central recycling facility near Johannesburg, where they are electrolytically stripped and steel and tin is recovered. Most of this steel and tin is used for reprocessing in the country; however, some of the recovered products are exported. For example, recycled steel from beverage cans with a higher tin content was exported to Pakistan to be used in lower grade steel manufacturing and briquetted, shredded cans were exported to Botswana to be used in cobalt smelting (Boroughs, 2008 and PACSA, 2011). Data on GHG emissions from steel manufacturing were sourced for the two most important steel manufacturers in the country (covering more than 90% of the production) from the Carbon Disclosure Project JSE Report (2011). The recycling factor calculated for steel is presented in Table 4.18.

Table 4-18: The calculation of a GHG emission factor for the recycling of steel in South Africa

Emission factor calculation	Source or reference
<u>Indirect, upstream processes</u> 1) Collection of steel (1 t of scrap with a rejection rate of 2% at sorting stage) Assuming 5-10 km one way carrying 100 kg (as done by local entrepreneurs) by personal car or small van 1 t*45 kg CO_2 e /tonne = 45 kg CO_2 e 2) Transport of steel Assuming 700 km in a truck (≥ 16 tonnes) 1 t *700km*0.190 kg CO_2 e / tonne / km = 133 kg CO_2 e ***Total 178 kg CO_2 e***	Reject of steel scrap obtained from Damgaard et al. (2009) Car emission factor obtained from Eisted et al. (2009). Truck emission factor obtained from Spielmann et al. (2004) cited in Eisted et al. (2009). Worst case scenario taken since South Africa has lower emission regulations for trucks (e.g. EURO 0, I and II, as opposed to EURO III, IV and V)
<u>Direct, recycling processes*</u> 1) Sorting 1 t*6.8 L diesel / tonne = 6.8 L diesel used by cranes 6.8 L* 2,7 kg CO_2 e / L diesel= 18.4 kg CO_2 e 2) Shredding 0.98t*50kWh/tonne=49 kWh 49kWh* 1.03 kg CO_2 e /kWh= 50.5 kg CO_2 e ***Total 68.9 kg CO_2 e***	Damgaard et al. (2009) Damgaard et al. (2009) Local electricity data (Eskom, 2011)
<u>Indirect, downstream processes</u> Savings from reprocessing and avoided virgin production of 0.98 t steel 0.98t*2890 kg CO_2 e/tonne= 2832.2 kg CO_2 e	Arcelor Mittal South Africa – data published in the Carbon Disclosure Project Report, 2011.
<u>Overall total</u> 178+68.9-2832.2= **-2586.9 kg CO_2 e**	

* electrolytical de-tinning not included due to lack of data

4.2.4.3 Recycling factors for plastics

Plastics make up about 12% of the municipal waste stream in South Africa (Friedrich and Trois, 2013). In 2010 an overall recycling rate of 18% was recorded and about 241 853 tonnes of plastics were recycled out of which 75.2% was plastics packaging. It was calculated that about

605 000 tonnes of plastic packaging was used in 2010 by the South African market, giving a recovery rate for plastic packaging of 30.1% (Plastics SA, 2012). Alone in 2009 about 52 000 tonnes plastics were picked directly off landfills, however, this amount declined slightly in 2010 (PACSA, 2011). There were 194 plastic recycling companies operating in 2010 in South Africa (Plastics SA, 2011).

Plastic recovered from municipal solid waste can be used for the production of new plastic and for energy recovery. In South Africa energy recovery from post-consumer plastics is currently not practised, but is under investigation (PACSA, 2011). Most of the recovered plastic is used for the production of other plastic products. For example, in 2009 the market applications for plastic recyclate in South Africa were: 30% for plastic film, 25% for injection moulding, 12% pipes, 11% for tapes fibres and filaments, 8% for toll and 8% for other smaller applications including timber replacement (polywood and wood composites 0.7% and profile, fencing poles and barricades 1%). Of the total plastics recovered 2% was exported, mainly to other Southern African countries (PACSA, 2011). It is estimated that about 4 840 permanent, formal jobs and about 34 500 informal jobs (waste pickers) have been created by the plastic recycling industry in 2009 (Plastics SA, 2011). The umbrella organisation for plastics in the country is Plastics SA (previously Plastic Federation of South Africa) and in plastics recycling a series of other organisations are involved (South African Plastics Organisation, Polystyrene Packaging Council, South African Vinyl Association, Polyco (representing polyolefins) and the PET Recycling Company). These organisations are funded by industry through a levy system which covers resin producer, brand owners, converters, bottlers and retailers (PACSA, 2011).

The main types of plastics are polyethylene (low (PE-LD), linear low (PE-LLD) and high density (PE-HD)), polyethylene terephthalate (PET), polypropylene (PP), polystyrene (PS) including expanded polystyrene (PS-E) and polyvinyl chloride rigid (PVC-U) and flexible (PVC-P) (PACSA, 2011). The different plastics types have to be separated prior to reprocessing and this adds more complexities to the recycling process. The most recent reported figures for the different plastic types recycled in the country are for 2010 when about 101 454 tonnes (43%) PE-LD, 39733 tonnes (16%) PE-HD, 39 855 tonnes (17%) PET, 38 614 tonnes (16%) PP, 11 929 tonnes (5%) PVC-P and 5% other plastics were recycled (Plastics SA, 2011). The plastics that were very little or not at all recycled in South Africa, in the year 2010, included multi-layer and metallised films, post-consumer PE-HD shopping bags, bioplastics, biodegradable and oxo biodegradable films, some cable insulation, PVC packing, engineering polymers (e.g. acetal, glass filled nylon and polycarbonate) and most of the expanded polystyrene post-consumer products (Plastics SA, 2011).

Most of the plastics recycled are used for the production of the same and/or other plastics and therefore, the recycling process is considered closed-looped. Recycled plastic in South Africa is used very little to substitute wood (only 1.7% in 2010) and not at all to substitute fuels (e.g. combustion instead of coal or oil in a cement kiln or in an energy recovery facility) as done internationally. The emission factor calculated for plastic recycling is presented in Table 4.19 and was performed for the 4 most recycled types of plastic, namely PE-LD, PE-HD, PP and PET, representing about 90% of the recycled plastic.

By using a weighted average based on the percentages of recycled plastic in South Africa, and the factors derived in Table 4.19 for each plastic type a mixed plastic factor of -980 kg CO_2 e per tonne was calculated for recycling of plastic from the South African municipal solid waste. It was assumed that the 5% categorised as other plastic had similar emissions as PE-HD (worst case scenario with regard to GHG savings).

Table 4-19: The calculation of a GHG emission factor for the recycling of plastics in South Africa

Emission factor calculation	**Source or reference**
Indirect, upstream processes 1) Collection of plastics (1 t of plastics with a rejection rate of 30% at sorting and washing stage) Assuming 5-10 km one way carrying 100 kg (as done by local entrepreneurs) by personal car or small van 1 t*45 kg CO_2 e /tonne = 45 kg CO_2 e 2) Transport of plastics Assuming 700 km in a truck (≥ 16 tonnes) 1 t *700km*0.190 kg CO_2e / tonne / km = 133 kg CO_2 e ***Total 178 kg CO_2 e***	Rejection rate obtained from Astrup et al. (2009) Car emission factor obtained from Eisted et al. (2009). Truck emission factor obtained from Spielmann et al. (2004) cited in Eisted et al. (2009). Worst case scenario taken since South Africa has lower emission regulations for trucks (e.g. EURO 0, I and II, as opposed to EURO III, IV and V)
Direct, recycling processes Washing, drying, compaction and granulation and final palletisation (30% loss) 0.7t*600 kWh/t*1.03kg CO_2 e/kWh = **432.6 kg CO_2 e**	Energy requirements obtained from Astrup et al. (2009). Worst case scenario assumed. Local electricity generation data used.
Indirect, downstream processes Avoided emissions due to substitution of virgin plastic for PE-LD 0.7t * 2100 kg CO_2 e/t = **1470 kg CO_2 e** PE-HD 0.7t* 1900 kg CO_2 e/t = **1330 kg CO_2 e** PP 0.7t* 2000 kg CO_2 e/t = **1400 kg CO_2 e** PET 0.7t*3490 kg CO_2 e/t = **2443 kg CO_2 e**	Unfortunately there is no local data on greenhouse gas emissions from plastics production. Data from PlasticsEurope (2012) for 2005, for a time horizon of 100 years were used. These are LCA inventory data from a cradle-to-gate perspective.
Overall total Following savings were calculated for: PE-LD 178+432.6-1470 = **- 859.4 kg CO_2 e** PE-HD 178+432.6- 1330 = **- 719.4 kg CO_2 e** PP 178+432.6- 1400 = **- 789.4 kg CO_2 e** PET 178+432.6- 2443 = **- 1832.4 kg CO_2 e**	

4.2.4.4 Recycling factors for paper

Paper and paper products represent about 18% of the municipal waste stream in South Africa (Friedrich and Trois, 2013) and consists mainly of printed paper and packaging. In 2010 the figures presented in Table 4.20 were consumed and recycled in the country.

Table 4-20: Tonnes of paper consumed and recycled in South Africa (Source: PAMSA, 2012)

Paper grades	Paper consumed	Paper recovered
Newsprint	272 360	143 102
Printing/writing	625 846	48 003 magazines and 189 204 office paper
Corrugating materials/containerboard	1 112 339	611 611
Other wrapping papers	96 719	-
Tissue	217 040	-
Other paper	27 613	98 279
Board	104 407	Included with corrugating materials
Total	**2 497 286**	**1 090 198**

From the paper consumed, it was calculated that 210 000 tonnes were exported with agricultural products and about 16 % (397 745 tonnes) are not suitable for recovery (e.g. tissue and sanitary products, cigarette paper, archive material etc.). Therefore, in 2010 in South Africa 1 878 163 tonnes of recoverable paper was consumed and 58% of it was recovered. In addition, in that year 10 208 tonnes of recycled paper was imported and 59 475 tonnes exported (PAMSA, 2012).

Paper recycling is done under the coordination of the Paper Recycling Association of South Africa which has 8 members, and it is a subsidiary of the Paper Manufacturers Association (PAMSA). The paper and cardboard recovered from municipal waste can be reprocessed as a mixed fraction or as a separate fraction (for different paper grades) depending on a variety of factors like separation, quality, etc. There are two main technologies for reprocessing the paper recovered, namely mechanical re-pulping and chemical-mechanical re-pulping. Merrild et al. (2009a) provide a description of these technologies, as well as the technologies used for the production of paper from virgin materials. Based on these data and local information obtained from PAMSA, Mondi SA, Sappi SA, a tissue paper and three packaging manufacturers, recycling factors were calculated for the different paper grades and these are presented in

Table 4.21. It should be noted that currently (2012) there is no closed loop recycling for white writing and printing paper. This paper grade is recycled into tissue paper. Because there are no local GHG data available on the emission for the production of tissues from virgin materials, it was assumed that the production of white, virgin writing and printing paper has the same emissions.

In recycling paper there is a material loss and a material quality loss. The material quality loss is specific for paper recycling and is explained by the fact that paper fibres get shorter with each reprocessing cycle and in order to obtain the same quality of paper new fibres have to be added. For the calculations presented in Table 9, a material loss of 10 % was taken, with the values in the literature (also supported by local data) ranging between 2 to 18 % (Merrild et al., 2009b). The material quality loss for packaging paper and cardboard, as well as for fine paper and mixed paper is cited in the literature as being 10% (Merrild et al., 2009b). This was only partially supported by local data, locally it seems these values are higher (10 to 20 % depending on individual paper plants). Therefore, a conservative overall loss of 20% was considered for paper recycling in South Africa. It should be noted that carbon sequestration by forests in relation to paper recycling was not taken into consideration in accordance to the approach by Schlamadinger et al. (1997) also used by Merrild et al. (2009a). This approach is justified in South Africa because wood for paper production in the country is sourced from forest plantations cultivated only for this purpose.

By using a weighted average based on the percentages of different recycled paper grades (see Table 4.20), and the derived factors for each paper grade as presented in Table 4.21, a mixed paper factor of - 568.5 kg CO_2 e per tonne was calculated for South African recycling of paper from municipal solid waste. It was assumed that all the paper under “other paper” in Table 4.20, representing about 9% of the recycled paper in the country, was recycled into packaging as corrugated materials and containerboard. It was assumed that office paper was recycled into tissue paper, and the recycled newsprint and magazines paper and into new newsprint paper.

Table 4-21: The calculation of a GHG emission factor for the recycling of paper in South Africa

Emission factor calculation	**Source, reference or conversion**
Indirect, upstream processes 1) Collection of paper Assuming 5-10 km one way carrying 100 kg (as done by local entrepreneurs) by personal car or small van 1 t*45 kg CO_2 e /tonne = 45 kg CO_2 e 2) Transport of paper Assuming 200 km in a truck (≥ 16 tonnes) 1,070t *200km*0.190 kg CO_2e / tonne / km = 40.7 kg CO2 e ***Total 85.7 kg CO_2 e***	Car emission factor obtained from Eisted et al. (2009). Truck emission factor obtained from Spielmann et al. (2004) cited in Eisted et al. (2009). Worst case scenario taken since South Africa has lower emission regulations for trucks (e.g. EURO 0, I and II, as opposed to EURO III, IV and V).
Direct, recycling processes Assuming a 0.2 material and material quality loss rate per tonne following GHG emissions have been calculated. Reprocessing white printing and writing paper into tissue paper 0.8*2142.7 kg CO_2 e/t = **1714.2 kg CO_2 e** Reprocessing newsprint and higher grades of paper into newsprint and magazine paper 0.8*755 kg CO_2 e/t = **604 kg CO_2 e** Reprocessing packaging and higher grades of paper into linerboard and corrugated medium 0.8*1425 kg CO_2 e/t = **1140 kg CO_2 e**	Material and material quality loss rate obtained from literature (see Merrild et al. (2009) for a summary) and from interviews with representatives from the local industry (PAMSA). GHG per tonne obtained from one local tissue manufacturer. GHG obtained per tonne from two major local mills and averaged. GHG data per tonne obtained from two major local plants and averaged.

Indirect, downstream processes Avoided emissions due to substitution of **virgin** paper 1) Printing and writing paper 0.8*2364.5 kg CO2 e/t = **1891.6 kg CO_2 e** 2) Newsprint 0.8*2875.1 kg CO_2 e/t = **2300 kg CO_2 e** 3) Packaging (corrugated materials, card- and containerboard) 0.8*2179.2kg CO_2 e/t = **1743.4 kg CO_2 e**	 Average emissions for this grade obtained through a weighted average calculation (based on production data) and the GHG product declaration from the two major companies in the country, covering 98% of local white paper production. Average emissions for this grade obtained through a weighted average calculation (based on production data) and the GHG product emissions provided by the two major companies in the country, covering 98% of local newsprint production. Average obtained through a weighted average calculation (based on production data) and the GHG product emissions (cradle to gate) provided by the three most important companies in the country, covering 50% of local production of packaging.
Overall total Following savings were calculated for: Tissue paper 85.7+1714.2-1891.6 = **- 91.7 kg CO_2 e / t** Newsprint 85.7+604-2300 = **-1610.3 kg CO_2 e / t** Packaging 85.7+1140-1743.4 = **-517.7 kg CO_2 e / t**	

4.2.4.5 Comparison of South African GHG emission factors for recycling with international factors

The GHG emission factors on recycling have been summarised and are compared with other factors from the literature. This comparison is presented in Table 4.22.

Table 4-22: Comparison of GHG emission factors developed for South Africa with recently published international factors (kg CO2 e /tonne recyclable)

Waste fraction/Material	Australia (RMIT, 2009) cited in UNEP (2010)	USA factors from EPA(2012)* as in WARM v12	Northern Europe - as summarized by various authors**	South Africa
Paper - mixed	-670 to -740	-3880 (-3.52)	390 to -4400 (Merrild et al., 2009)	-568.5
Plastics – PE-HD	Mixed plastics 0 to -1180	-884.9 (-0.86)	-55 to -1595 (Astrup et al., 2009)	-719.4***
Plastics - PET		-1223.6 (-1.11)	Not specified	-1832.4
Glass	-560 to -620	-309 (-0.28)	-500 to -1500 (Larsen et al., 2009b)	-290.1
Ferrous metal (steel)	-400 to -440	-1984 (1.80)	-560 to -2360 (Damgaard et al., 2009)	-2586.9
Aluminium	-17720	-7683 to -9800 (-6.97 to -8.89)	-5040 to -19340 (Damgaard et al., 2009)	-19110.7

*original values (EPA, 2012) expressed in $MTCO_2E$/ton (US) are presented in brackets

**all of these authors exclude transport in the indirect upstream processes

*** the factor for mixed plastics in South Africa is - 980 CO_2 e / t

Following international trends, the recycling of all waste materials investigated show GHG savings. However, as seen from Table 10 the recycling of some waste components (e.g. aluminium and steel) have much higher GHG savings in South Africa as compared to other parts of the world. This is due to the high carbon intensity of these processes and the use of electricity, derived from coal, for the production of virgin materials that these components replace when recycled. These results underline the interdependency of the waste management processes and the energy system in which they operate, with South Africa as a typical example of carbon intensive energy dependency. Many developing countries, most notably China and India, are also in this position. The emission factors calculated also highlight that when considering GHG from recycling, and in general from waste management a holistic, life cycle approach has to be used to reflect best the overall emissions or savings from waste management processes.

As stated in the Integrated Resource Plan 2010-2030 (RSA – Department of Energy, 2011) South Africa aims to decrease its coal dependency and to increase the proportion of energy from renewable resources. Therefore, in long term the GHG emissions for electricity are expected to decrease. These changes have to be taken into account when calculating GHG emission factors for recycling municipal solid waste and these factors should be revised periodically to reflect progress with regard to the use of renewable resources.

4.2.5. The development of GHG emission factors for composting

Composting of the organic fraction of the municipal waste is practised on a large scale in Europe and the USA. However, it is practiced only by a minority of municipalities in South Africa (e.g. City of Johannesburg, City of Cape Town, eThekwini Municipality) and even some of these municipalities perform it only at experimental/pilot study scale (i.e. eThekwini Municipality). There is, however, an increased interest in composting and one project (Klipheuwel – Cape Town) is applying for CDM accreditation. From the technologies presented in the literature (see Boldrin et al., 2009 for a summary) South African municipalities use windrows and piles (static and aerated) in an open environment. Cooperative composting by community groups, as well as home composting is also practiced, but there is little information on these projects.

With regard to GHG emission, composting as a process causes net contributions but also has the potential to bring about GHG savings. GHG are released due to the energy needed by the composting facility (i.e. machinery used) and by the degradation process itself which produces small amounts of methane and nitrous oxide. GHG savings due to compost occur when compost is used instead of an inorganic fertilizer and when carbon is bound to the soil after compost was applied (Boldrin et al., 2009). The literature shows that compost can replace peat in the production of growth media and there are GHG savings derived from this process (Boldrin et al., 2009). However, in South Africa this is not practised and the main uses for compost from municipal waste are for horticultural and agricultural purposes (by homeowners and organic farmers) and for landscaping to a lesser degree (Ekelund and Nyström, 2007). In this context, only the savings from replacing inorganic fertilizer and from the binding of carbon to the soil could be taken into account when calculating an emission factor for composting.

The calculations involved and the factor derived are presented in Table 4.23. Mass balance calculations were used to determine how much of the carbon and nitrogen in the initial organic waste (i.e. garden waste) was present in the resultant compost and how much was emitted as GHG (CO_2, CH_4 and N_2O). The characterisation tests undertaken at the University of KwaZulu-Natal, Durban, South Africa (Iyilade, 2009; Plüg, 2009 and Moodley, 2011) showed that the garden waste had, on average, a carbon content of 48.6 % (dry waste) and a nitrogen content of 0.63 % (dry waste). The average moisture content was 40%. Therefore, on average, one tonne of wet garden waste contained 292 kg of carbon and 3.8 kg of nitrogen. The carbon loss was calculated as 77.3 % for the turned windrow technology and as 70.5 % for the aerated dome technology. The methodology presented by Boldrin et al. (2009) was followed for calculating the GHG from composting using the two technologies employed (i.e. turned windrows and aerated dome - both in an open setting). It was assumed that 2.7 % (one of the worst case scenarios) of the degraded carbon will become CH_4 and 1.8 % of the input nitrogen will result in the formation of N_2O (Boldrin et al., 2009). The carbon content in the compost on a dry basis was 29.04 % for the turned windrow and 22.04 % for the dome aeration technology. The nitrogen content, also on a dry basis, was 1.65% (turned windrow) and 0.96% (aerated dome). The moisture content of the compost was 60% for the windrow compost and 55% for the dome aerated compost. Taking into account the moisture content of the compost, on a wet basis, one tonne of turned windrow compost contains 116.16 kg carbon and 6.60 kg nitrogen and one tonne of dome aerated compost contains 99.18 kg carbon and 4.32 kg nitrogen. Phosphorus and potassium in the compost were unfortunately not analysed and values from the literature had to be used in calculations.

Table 4-23: The GHG emission factor for composting 1 tonne wet garden waste in South Africa

Emission factor calculation	**Source, reference or comment**
Indirect, upstream processes Transport of garden waste to composting facility Assuming 30 km distance in a truck* 1 t *30km*0.190 kg CO2e / tonne / km = 5.7 kg CO_2 e	Truck emission factor obtained from Spielmann et al. (2004) cited in Eisted et al. (2009). Worst case scenario taken since South Africa has lower emission regulations for trucks (e.g. EURO 0, I and II, as opposed to EURO III, IV and V)
Direct GHG emissions from the composting processes 1. Combustion of diesel for shredding, loading and turning the garden waste/compost 1 tonne * 3 diesel *2,7 kg CO_2 e /L = 8.1 kg CO_2 e 2. Methane (CH4) release due to composting Turned windrow technology 292 kg C*0.773*0.027*16/12 = 8.13 kg CH_4 8.13 kg CH_4 *25 kg CO_2 e / kg = 203.25 kg CO_2 e Dome aeration technology 292 kg C*0.705*0.027*16/12 = 7.41 kg CH_4 7.41 kg CH4 *25 kg CO2 e / kg = 185.25 kg CO_2 e 3. Nitrous oxide release due to composting For both technologies 3.8 kg N*0.018*44/28 = 0.11 kg N_2O 0.11 kg N_2O * 298 kg CO_2 e / kg = 32.78 kg CO_2 e Total Turned windrow 8.1+203.25+32.78 = ***244.13 kg CO_2 e*** Dome aeration 8.1+185.25+32.78 = ***226.13 kg CO_2 e***	Value for diesel consumption obtained from Boldrin et al. (2009) In addition to the GHG presented, biogenic CO2 was released due to composting as follows: Turned windrow 292 kg C*0.773*0.973*44/12 = 805.3 kg CO_2 Dome aeration technology 292 kg C*0.705*0.973*44/12 = 734.5 kg CO_2
Indirect, downstream processes 1. Savings due to the substitution of inorganic fertiliser due to compost application Assuming a substitution ratio for N of 50 %, the 3.76 kg of N resultant from 1 tonne of garden waste composted through turned windrows would replace 1.88 kg of nitrogen from an inorganic fertilizer. Similarly 3.75 kg of N resultant from 1 tonne of garden waste composted through aerated dome technology will replace 1.87 kg of nitrogen from an inorganic fertiliser. Assuming that 13 kg CO_2 e are emitted per kg N, 3.095 kg CO_2 e per	Substitution ratios for N summarised by Boldrin et al. (2009) is 20 to 60%. Worst case scenario from the literature

kg P and 1.53 kg CO2 e per kg K the production of inorganic fertilisers the savings are as follows: Turned windrow: 1.88 kg N*13 kg CO_2 e = 24.44 kg CO_2 e 0.6 kg P * 3.095 kg CO_2 e = 1.86 kg CO_2 e 3.5 kg K * 1.53 kg CO_2 e = 5.36 kg CO_2 e **Total 31.66 kg CO_2 e** Aerated dome technology 1.87 kg N * 13 kg CO_2 e = 24.31 kg CO_2 e 0.6 kg P * 3.095 kg CO_2 e = 1.86 kg CO_2 e 3.5 kg K * 1.53 kg CO_2 e = 5.36 kg CO_2 e **Total 31.53 kg CO_2 e**	values summarised by Boldrin et al. (2009) in Table 4-19 was taken, because of the coal dependence of South African electricity, which is used in the manufacturing processes for inorganic fertiliser. P and K values per tonne of garden waste composted taken from Boldrin et al. (2009).
Spreading of compost on land** requires 0.19 to 0.40 L diesel per tonne of garden waste, therefore at the most 0.40 L * 2.7 kg CO_2 e / L = 1.08 kg CO_2 e are released. 2. Savings due to carbon bound to soil due to compost application	Boldrin et al. (2009)
Assuming that 8% of the carbon in the compost is bound to soil (100 year time frame) following savings are calculated: Turned windrow compost 116.16 kg C * 0.08 * 44/12 = 34.07 kg CO_2 e Dome aerated compost 99.18 kg C * 0.08 * 44/12 = 29.09 kg CO_2 e Total Turned windrow 31.66- 1.08 + 34.07 = 64.65 kg CO_2 e Aerated dome 31.53 -1.08 + 29.09 = 59.54 kg CO_2 e	% C bound from Boldrin et al. (2009)
<u>Overall total</u> Turned windrow 5.7+244.13-64.65=**185.18 kg CO_2 e** Aerated dome 5.7+226.13-59.54=**172.29 kg CO_2 e**	

*if the current situation changes and garden waste is collected on a large scale and separately from households, this assumption has to change. At the moment garden waste is transported from collection points (where households bring it) and from nurseries, garden centers, etc. The garden waste from households represents a small percentage of the total garden waste collected and composted in the local municipalities. The emissions due to the transport of this waste by households to collection points have been left out for the time being.

**Emission of N_2O after spreading on the land of compost was considered zero.

An emission factor has been calculated only for garden waste composting, because at the moment in South Africa in many municipalities, this waste is separated from the municipal steam. This is not the case for food waste and there are no immediate plans to promote such separation.

When comparing the GHG emissions from composting calculated for South Africa with similar figures from the literature, it can be observed that the South African factors are positive, meaning that the process has net GHG emissions. Boldrin et al. (2009) present a range of emissions for Europe for composting, varying from -900 (net savings) to 300 (net emissions) kg CO_2 e per tonne of wet waste. The South African emissions are within this range. Other published results show savings of – 220 kg CO_2 e per tonne of wet waste in the USA (EPA, 2012) and Barton et al. (2008) consider composting to be carbon neutral in developing countries.

4.2.6. Uncertainty and limitations – A brief overview

For the calculation of the GHG emission factors presented in the sections above, numerous assumptions had to be made. Each of these assumptions might have introduced a margin of error. From the emission factors calculated for recycling, the one factor derived from the least number of assumptions and the most measured parameters is the emission factor for glass. The ones with the most assumptions used are the GHG emission factors for the different plastic types. In particular, the use of European data on plastic production when calculating GHG from substitution of raw plastics is an underestimation of the likely South African emissions, which means in reality GHG savings are higher. This is because the South African electricity mix is different from the European one and has a higher GHG burden. Electricity is used in the production of all types of plastics.

Representativeness of compost calculations is another limitation which has to be noted. Composting studies rely on specific parameters derived from individual studies which then are generalised for a waste management system of a country and region. This is the case also for the South African situation, in which composting has been studied for garden waste from the eThekwini Municipality, which is situated in a subtropical location. The vegetation and the other parameters (most importantly temperature and moisture) might not be representative for other parts of the country, which have arid conditions. However, until more detailed composting studies are done, covering more climatic areas in the country, the factors calculated in this paper are considered more representative than the theoretical, international ones published in the literature.

4.2.7. Conclusions

This paper is part of a larger study which investigated and developed GHG emission factors for waste management processes in South African municipalities. As such, emission factors for recycling and composting were calculated. This is important for South Africa since it will enable municipalities which have little data to perform calculations on their GHG emissions from waste management. The results for recycling of municipal solid waste show that all recycling processes bring about net GHG savings. These savings range from -290 kg CO_2 e (glass) to – 19111 kg CO_2 e (metals - Al) per tonne of recyclable. Most of the GHG factors calculated for South Africa are in the ranges presented in the literature, with the exception of energy intensive materials (i.e. aluminium and steel) where the GHG savings due to recycling are higher in South Africa. These results underline the interdependency of the waste management system of a country/region with other systems, most importantly the energy system, with South Africa having a carbon intensive energy generation system dependent on coal. As this dependency is reduced and more renewable resources will be used in the future for energy generation, the GHG recycling factors should be revised to reflect these changes.

Composting of garden waste as it was done in the few studies undertaken at UKZN in Durban, has net GHG emissions (about 185 kg CO_2 e per tonne of wet garden waste for turned windrow and 172 kg CO_2 e per tonne of wet garden waste for the aerated dome technology). These results are also within the ranges presented in the literature. However, more research is needed to refine these figures for more arid parts of the country which have large municipal areas (e.g. Johannesburg) and - which are important waste generators.

The emission factors developed in this study are targeted towards waste managers and professionals involved with decision making affecting the municipal waste management system in local municipalities. The use of these factors will enable scenario modelling and local choices for best practices with regard to reducing GHG emissions from municipal waste management. Therefore, in a next step a simple tool should be developed and distributed to local authorities. It should facilitate the use of these factors for GHG calculations and accounting at this level, since it is the most important level for operational decision making, with significant outcomes in terms of GHG savings from waste management in South Africa. In addition, an educational tool for households, schools and other general institutions should also be developed based on these emission factors.

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Chapter 5 Current and Future GHG Emissions from the Management of Municipal Solid Waste in the eThekwini Municipality – South Africa

5.1 Abstract

When considering projections of GHG emissions from waste there is a paucity of such studies for municipalities in developing countries and particular in Africa. This study addressed that gap and by taking a life cycle approach, current and projected emissions for the eThekwini Municipality situated in South Africa are calculated. In 2012 (current situation) the GHG emissions from the entire waste management system in this municipality were in fact net savings of about -161780 tonnes CO_2 e (CO_2 equivalents). This is mainly due to landfill gas to electricity CDM projects and due to recycling in the municipality. The closure of one of the three local landfill sites in 2014 and the re-directioning of the majority of waste to another landfill site which does not have LFG collection and utilisation, will cause an increase of GHG emissions to 294670 tonnes CO_2 e. Future projections for the year 2020 have been made and, depending on the waste technologies used different savings can be achieved from waste management in the municipality. An increase in recycling and the introduction of anaerobic digestion and composting has the potential to significantly reduce GHG emissions from waste. However, only the introduction of LFG collection and utilisation systems will result in the highest possible overall GHG savings. The results presented in this paper show that life cycle based GHG emission factors for waste and their use can support decision-making for municipalities in the local context and aid in optimising the waste management system as to achieve the highest possible GHG savings.

5.2 Introduction

5.2.1 Context and objectives of the study

Human caused climate change is one of the most important environmental issues of our times and the increase in atmospheric emissions of GHGs is a cause of concern and action. South Africa is the 12th largest emitter of GHGs globally. An estimated 387 million metric tonnes CO_2 e. were released in 2004, representing 1.6 % of global emissions (RSA National Treasury, 2010). About 2 % of the South African GHG emissions are due to the waste sector (Greenhouse gas inventory, 2009) and globally this sector accounts for about 3 % (Bogner et al., 2008). In this context, the calculation of current emissions and future projections of GHG for waste management, are performed with increasing frequency by many organisations, at different geographical and operational levels (e.g. Moni et al., 2006; Mohareb et al., 2008; Bakas et al., 2011). A few of these studies also investigate developing countries, but they focus on current emissions (e.g. Liamsanguan and Gheewala, 2008 and Zhao et al., 2009). Detailed future projections of GHG emissions have not been performed for these countries and in particular for countries in Africa. Therefore, this study aims to fill this knowledge gap in the South African context and presents current emissions and future projections of GHGs from municipal solid waste management for one metropolitan area in the country, namely, the eThekwini Municipality. In addition it investigates consequences of planned changes in the current waste disposal system and possible future waste management scenarios which can lead to lower GHG emissions feasible in the local context.

5.2.2 Waste management in the eThekwini Municipality, South Africa

The eThekwini Municipality is located on the eastern coast of South Africa, within the Province of KwaZulu-Natal. The core city of this municipality is Durban, which has the largest port in the country and the subcontinent. The municipality has a population of about 3.5 mil people and covers an area of 2297 km^2, with a coastline of 98 km along the Indian Ocean (eThekwini Municipality IDP, 2012). Inland, Durban is surrounded by other urban nodes, as well as other more sparsely populated peri-urban areas. The number of households in the municipality is about 956000 - of which about 55% are formal (flats and houses), 34% informal (shacks and backyard dwellings) and about 11% are rural (traditional clusters and formal houses) (eThekwini Municipality IDP, 2012). In the informal and rural areas there are backlogs in the provision of infrastructure and services (i.e. housing, roads, electricity, potable water and sanitation). The Community Survey of 2007 showed that 88.5% of the households in the municipality had their waste collected by the municipality, 1.4 % used a communal dump, 7.3 % their own dump, 2,5 % had no refuse collection and disposal and 0.3 % used other means. Since then waste collection has been extended to cover, in one form or another, all households in the municipality (eThekwini Municipality IDP, 2012). However, the waste management system of this municipality still faces many problems inherent in developing countries (Matete and Trois, 2008).

Durban Solid Waste (DSW) is the municipal waste managment unit and operates 3 active sanitary landfill sites (Bisasar Road, Mariannhill and Buffelsdraai landfill sites), 23 recycling and garden refuse drop-off centres, 6 major transfer stations, 2 landfill-gas to electricity plants (operated as CDM projects) and 2 leachate plants (Durban Solid Waste, 2012). Figure 5.1 illustrates the geographical location of waste management infrastructure in the municipality. The largest of the 3 landfill sites (Bisasar Road) is planned to be closed in 2014 and the waste will be diverted to the two remaining landfill sites, with the bulk (90-95%) planned to be landfilled, in the short term, in the Buffelsdraai landfill site. Since this landfill site is situated in the northern parts of the municipality, transportation of waste will increase by about 40 km. Builders rubble will still be accepted at the Bisasar Road landfill site after the closure and there are plans for diverting garden waste towards composting and increasing recycling. There are also plans to develop a regional landfill site within the western parts of the municipality, however, at this stage (2013) the environmental impact assessment for this regional site is awaiting approval. In terms of waste collection and management, the levels of service differ between different areas of the municipality. In the formal areas there is a regular collection service provided (usually once a week) for domestic waste (which is collected in black bags), recyclables (currently paper and plastics collected in orange bags) and garden refuse (collected in blue bags) for which households are charged. For the informal settlement areas, limited collection services are provided at defined drop-off points free of charge. DSW also provides street cleansing and verge maintenance for all the roads of the municipalities. The municipal solid waste is collected by a fleet of about 151 vehicles operated by DSW, but small contractors are also involved in the collection of municipal solid waste, mainly in informal areas (where access of large vehicles is restricted) and in the kerbside collection of recyclables.

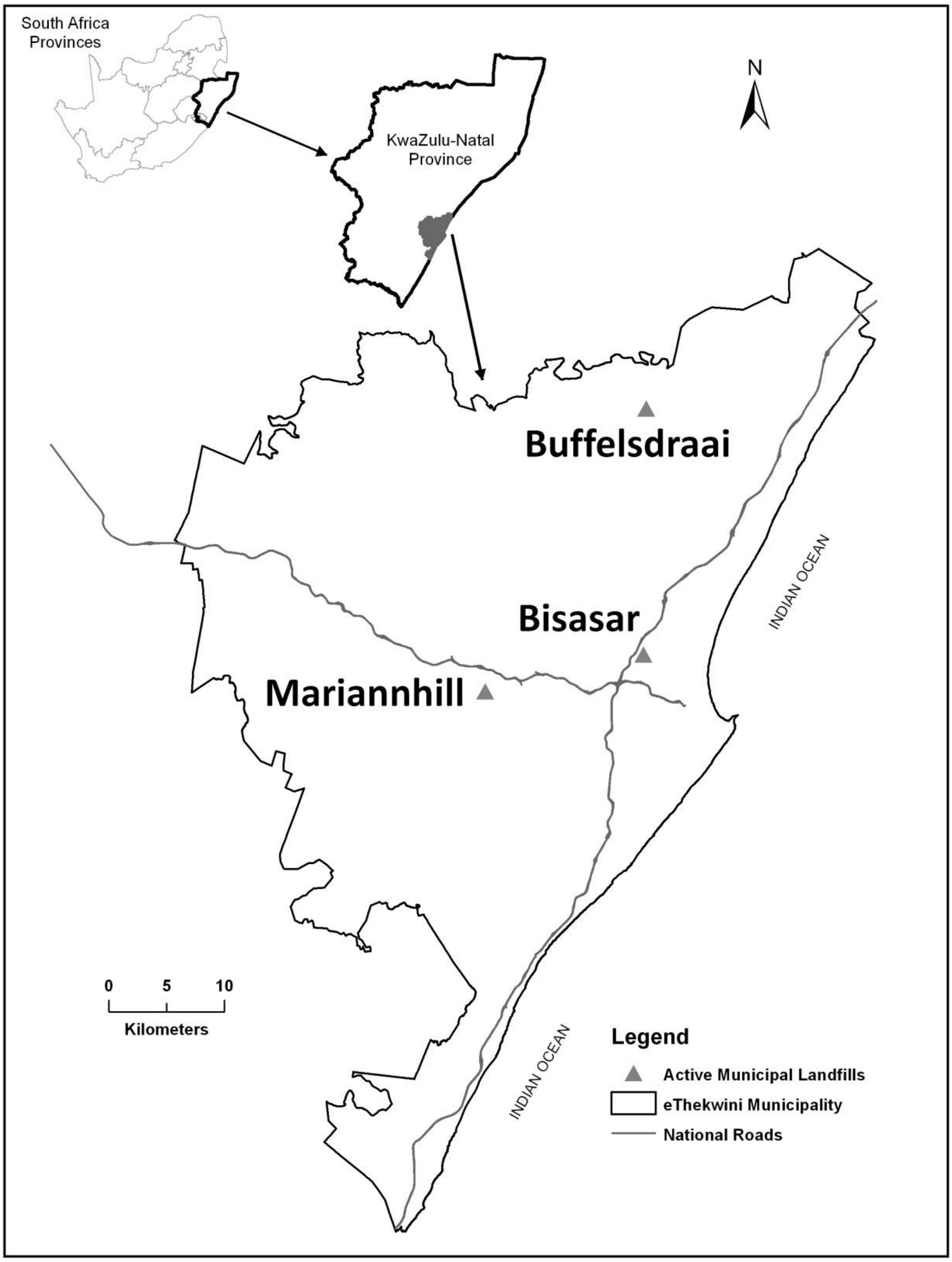

Figure 5-1: eThekwini Municipality Landfills

In addition to DSW, which is a municipal entity, in the eThekwini Municipality there are several other private companies involved with collection and disposal of waste, but only two of these private companies (Wasteman and Enviroserv) own landfill sites. One of these privately owned landfill sites had been recently closed in November 2011. These private companies cater mainly for industrial customers which pay for this service, but do accept limited amounts of general municipal (commercial and industrial) waste. In 2011 about 225 000 tonnes of waste were disposed by these two companies, representing about 13 % of all municipal waste. Due to the closure of one site this amount decreased in 2012, however, exact data were not finalised. Because the amount disposed in the one remaining private landfill is relatively small and because of lack of data on the composition of this waste, this landfill site was not included in this study.

Recycling of household and business waste is undertaken in the municipality via three routes, namely kerbside collection, buy back centers and drop off centers. There are designated drop off centers placed strategically in commercial and high income areas, however, each of the DSW garden refuse drop off sites also accepts recyclable materials. Buy back centers are targeting low-income groups, which use recycling for income generation. In certain areas there are chains for the handling of recyclables, with persons owning transport buying from primary collectors and selling it on further. Kerbside collection of recyclables (initially paper and plastics, but to be extended for other materials) was initiated in 2007 by DSW in collaboration with a local paper recycling company. Since then, the initiative has been extended to cover most formal areas in the municipality and is run successfully as a separate collection service scheduled to match the general waste collection. This simplifies participation from households, since general waste and recyclables are put out once a week on the kerbside. The mixed recyclables are transported to a central private material recovery facility (MRF), where they are separated manually and sold on.

Garden waste is separated in formal households and placed in blue bags for collection. This is done for financial reasons - blue bags costing the households more. However, during collection the garden waste bags (blue) are mixed with the general waste bags (black). In addition there are 14 drop-off centers for garden waste and this waste is separated. Also the garden waste that is generated by the Parks Department of the municipality is brought to landfills as separated waste. This separated garden waste is currently landfilled. Centralised, large-scale composting is not practiced currently by the eThekwini Municipality, although a few pilot initiatives have been undertaken in the last decade. Incineration of municipal solid waste is also not used in the eThekwini Municipality and not in South Africa. There are incidents of illegal back-yard burning of waste, but these seem not to be wide spread in the municipality. However, there is a lack of data and research for the municipality and for the country as a whole on back yard burning as well as on home composting. Therefore, home composting and incineration have not been included in this study.

5.3 Methodology

The aim of this study was to generate information on the GHG emissions due to the current waste management practised by the eThekwini Municipality and to estimate future emissions from this system. Some of these emissions, namely those from municipal landfill sites, have been quantified and trends for the future have been established under the CDM projects and these quantified emissions have been included in the municipal GHG inventories (Friedrich and Trois, 2010). However, other emissions like those from collection and transport of municipal solid waste and from recycling have not been previously quantified. Secondly, in addition to generating GHG information, this study aims to investigate possible waste management scenarios which might mitigate overall GHG emissions from solid waste management in the municipality. Thirdly, the different GHG short-term consequences from closing down the most important landfill site operated in the municipality (i.e. Bisasar Road will be closed down in 2014) were investigated. To achieve these aims different quantification methodologies for GHG emissions from municipal waste were investigated.

The quantification of GHG emissions from municipal solid waste can be undertaken by following five methodologies as presented by Kennedy et al. (2009). These methodologies are based on two approaches if the time factor is considered for the anaerobic degradation of waste in landfills and the associated release of GHGs (mainly methane). In the first type of approach the carbon landfilled is viewed per total and the time needed for the release of GHG emissions is not considered. The calculation of such instantaneous GHG emissions from landfill sites is called theoretical yield gas method (IPCC, 1996) or the methane commitment method (Mohareb et al., 2011). This approach uses the degradable carbon content of the waste landfilled, together with certain dissimilation pathways and parameters to calculate how much of this carbon will end up as methane, irrespective when in time this methane will be emitted. The second type of approach takes into account the time factor in the degradation of waste and the generation of methane emissions and a series of models have been developed to predict how much and when these methane emissions will occur. Kamalan et al. (2011) and Thompson et al. (2009) present reviews of such models. This approach is called the waste in place model. A particular application of this approach (the GasSim model) was used for the quantification of GHGs (mainly methane) from municipal waste for the local municipality for the CDM calculations performed and for the GHG inventories generated (Friedrich and Trois, 2010). However, there are several shortcomings associated with this approach, in particular its focus only on landfills and its associated inadequacy in comparing GHG emissions from different individual waste management processes. Therefore, the methodology chosen for this study was a life cycle based quantification using local GHG emission factors. This falls under the broad approach of a theoretical yield gas method (also called methane commitment method). This approach is used for planning in many parts of the world and local GHG emission factors and associated electronic tools have been developed (see Friedrich and Trois, 2013a for a review).

5.3.1 General approach

The general approach of the study is based on a simplified life cycle assessment framework which uses GHG emission factors developed for waste management in the South African context (Friedrich and Trois, 2013a and Friedrich and Trois, 2013b). The functional unit used for analysis is the disposal of municipal solid waste in the eThekwini Municipality for a given year. The most recent data shows that in 2012 the three municipal landfill sites received about 1.5 million tonnes general waste consisting of waste from households, businesses, as well as construction and demolition waste and tyres as shown in Table 5.1.

Table 5-1: Tonnes of waste disposed at the 3 municipal landfill sites (LFS) in 2012 (Source: DSW, 2013)

Type of waste	Bisasar Rd LFS	Mariannhill LFS	Buffelsdraai LFS	Total
DSW (about 80% household waste)	445188	105387	43803.4	594378.4
Commercial/industrial general waste	106528	46873.5	7810.1	161211.6
Garden Refuse (separated)	34341.5	8836.8	1538.1	44716.4
Builders Rubble	88740.8	15770.9	9359.3	113871
Mixed Loads	13592.1	2005.8	768.2	16366.1
Tyres	497375	49797.5	10393.6	557566.1
Light type refuse	458.3	1088.4	4.4	1551.1
Others	205.2	218.4	22.9	446.5
Total excluding tyres and builders rubble	620330 (73.52%)	168368 (19.96%)	54998 (6.52%)	843696 (100%)
Overall total	1209902	250215	74766.8	1534884

5.3.2 Data collection, calculations and assumptions

Data on the waste management processes investigated in this paper were obtained from published sources, as well as through interviews, site visits and direct observations. Table 5.2 presents the types of data used and their sources, as well as some observations regarding data quality.

Table 5-2: Data types and sources

Data types	Data sources	Observations
Waste composition	Haultec, 1998 Mgingqizane, 2004 Machetti, 2007 Purchase, 2007 Douglas, 2007	A mixture of waste composition data. Some of it is outdated.
Waste quantities	DSW	Up to date, audited data obtained for the amounts of solid municipal waste received by each landfill site.
Emission factors	Friedrich and Trois, 2013a and Friedrich and Trois, 2013b	Locally developed, up to date, emission factors used.
Recycling rates	PACSA Paper Recycling Association of South Africa The Glass Recycling Company	Most recent recycling rates used. Projected recycling rates for 2020 have uncertainties associated with them.

The modelling of GHG for future scenarios was based on a predicted amount of waste generated. This amount was calculated with the help of a regression analysis (best fitted line) based on the amounts generated and reaching municipal landfills in the period 2008 to 2012 (5 years). The R squared value of this regression is 0.766, meaning that the regression line fits the set of data reasonably well. This analysis showed that the amount of municipal solid waste (excluding tyres and construction and demolition waste) will increase from 843696 tonnes recorded in 2012 (see Table 1) to about 897407 tonnes in 2014 and to 1117671 tonnes in 2020. For 2012 for a 95% confidence limit, the lowest value was 708372 tonnes and the highest value was 939599 tonnes. For 2020 for a 95% confidence limit the lowest value was 1002057 tonnes and the highest value 1233284 tonnes. Since the timeframes (2 and 7 years) for the predicted waste amounts is relatively short, a constant margin of error was used. If amounts of waste are to be predicted for timeframes extending longer in the future more detailed time series modelling should be undertaken. It was also assumed that the composition of the waste will remain the same for the next 7 years.

A detailed waste composition analysis for the eThekwini Municipality was undertaken only once, in 1998 (Haultec, 1998) and is outdated. The results have been updated by Douglas (2007) by averaging the 1998 values with those from more recent studies (Mgingqizane, 2004; Marchetti, 2007) which sampled municipal waste on a smaller scale in once-off exercises in different areas. The municipal average calculated is presented in Table 5.3. Unfortunately, no statistical analysis was undertaken for the updated average. This average is very close to the calculated average for the South African solid waste composition (Friedrich and Trois, 2013a – see Table 3) and therefore, the GHG emission factors calculated for South Africa by the same authors based on this average, have been considered appropriate for this study. It should be noted that waste composition has been identified as one of the significant uncertainties for LCA studies and implicit carbon calculations (Slagstad and Bratebø, 2012) for countries and municipalities in the developed world, where more data and statistical analyses are available for waste management.

Table 5-3: Waste compositions (percentages)

Waste fraction	**Updated eThekwini overall municipal average (Douglas, 2007)**	**South African municipal average (Friedrich and Trois, 2013a)**
Organic	28.22	26.0±2.6
Green (incl. wood)	18.00	18.2±1.44
Plastics total of which 1. LDPE 2. HDPE 3. PET 4. Mixed plastics	11.95 3.13 1.45 1.64 5.73	12.1±0.45
Paper and cardboard total of which 1. White paper 2. Mixed paper 3. Cardboard	16.12 1.62 3.75 10.76	18.2±0.65
Glass	7.21	6.9±0.54
Metals	3.46	3.9±0.35
Other	15.03	14.7±3.35
Total	100	100±3.35

The amounts of waste presented in Table 5.1 and the emission factors as summarised in Table 5.4 provided the basis for the calculations for the different solid waste management scenarios modelled for the municipality. These emission factors have been derived through an LCA approach and all the waste management processes were taken into account (i.e. collection, transport, handling, treatment, disposal and/or re-use and recycling). Detailed calculations are presented in Friedrich and Trois (2013a and 2013b). An emission factor of -271,8 kg CO_2 e /

tonne organic municipal waste, was used for anaerobic digestion (AD), as derived by Jagath and Trois (2011) for the local context. The emission factors used in this study have been derived in accordance to international practice in the field and can be compared with international factors calculated for the USA and Europe (e.g. WARM, 2012; Manfredi et al., 2009) The emission factors for municipal solid waste presented in the literature are calculated based on the assumption that all GHG emissions from waste management take place instantaneously.

Table 5-4: GHG emission factors per tonne of South African wet waste/recyclable (kg CO2 e / tonne) (Source: Friedrich and Trois, 2013a and Friedrich and Trois, 2013b)

Waste Process/Technology	**Emission factor**
Collection and transport	11.3
Landfilling*	1016.3
Landfilling* with gas collection and flaring	101.2
Landfilling* with gas collection and electricity generation	-144.5
Composting garden waste	185
Recycling	
1. Paper – mixed	-568.5
2. Paper – office (white)	-91.7
3. Paper – newsprint	-1610.3
4. Paper – packaging (cardboard)	-517.7
5. Glass	-290.1
6. Metals – Al	-19110.7
7. Metals - Fe	-2586.9
8. Plastic – PE-LD	-859.4
9. Plastic – PE-HD	-719.4
10. Plastic – PP	-789.4
11. Plastic – PET	-1832.4
12. Plastics - mixed	-980

*Carbon storage (i.e. sequestration) included

This study presents more accurate calculations of GHG emissions from waste, because of the use of locally developed emission factors for processes included in the waste management system of a major South African municipality. Previously, for this type of calculations international emission factors (from USA and Europe) were used, resulting in increased uncertainties. The use of these factors in this manner provides a consistent framework and format to compare waste management processes and alternatives. It also makes it possible to include waste management processes not previously researched from a GHG point of view (e.g. transport and recycling).

5.3.3 Description of scenarios

GHG emission calculations from the management of municipal solid waste have been performed for the year 2012 (considered current scenario), for the year 2014 when the Bisasar Road landfill site will close as the entire waste management system of the municipality will change substantially and for 2020 because it allows for an improvement analysis and can guide the planning of the new waste management system.

5.3.3.1 Current scenario with and without LFG collection and use

The **current (2012) scenario** relies on the disposal in the 3 existing municipal landfills (Bisasar Road, Mariannhill and Buffelsdraai) as presented in Section 5.2.2. Two of these landfill sites have LFG collection and electricity generation (Bisasar Road and Mariannhill) and the third, most recently established one (Buffelsdraai) does not. There is no composting in this scenario but recycling as currently undertaken is included. Table 5.5 presents the percentage recycled in 2009/2010, as well the targets for recycling for the next few years. This table was updated for 2012 utilising recycling rates published by the various sources (industry federations and Packaging Council of South Africa (PACSA)). Obtaining detailed data on municipal solid waste recycling in the eThekwini Municipality was attempted for collection, but unfortunately, because of the competitive and fragmented nature of recycling companies and the subsequent reluctance to divulge data on quantities recycled locally, the collection of these data was only partially successful. Therefore, the quantities recycled were calculated based on the waste composition, the amounts of waste reaching the municipal landfills and the recycling rate for each material (paper, metals, glass and plastics) reported at national level. To check the accuracy of values obtained, additional calculations were done for all recyclables by using national figures, as reported by industry associations and the assumption that recycling in the eThekwini Municipality follows the same quantitative trends, based on income (9 % of the national income is generated in this municipality, therefore 9 % of the national figures reported for glass, paper, metals and plastics were attributed to recycling in this municipality). This method of calculation of what is recycled by a municipality is extensively used by the packaging recycling industry in the country (Harper, 2013). For paper, glass and plastics the calculations based on income were within 2, 3 and 10 %, respectively, compared to the amounts calculated based on the waste composition and the respective tonnages received by landfills (i.e. the percentages not recycled ended up in the landfill and the percentages recycled and reflected by each recycling rate were previously diverted). For metals the data obtained was only for metal used in packaging (148 282 tonnes recycled in 2011 in South Africa) and it was not possible to make meaningful comparisons, since metals from many sources besides packaging are recycled and diverted from landfills by households and particularly industry. Another significant assumption with regard to metals is that all metals are considered to be ferrous based (steel) since it was not known what the actual metal spilt is. However, this provides a worse-case scenario with regard to GHG savings from metal recycling.

Table 5-5: Percentages recycled for 2009 and 2012 and future recycling targets in South Africa (adapted from PACSA, 2011)

	2009 Recycling rates	2012 Recycling rates*	Year 2014 target by PACSA (2011)	Potential future rates for 2020**
Paper	56%	59%	61%	66 to 71%
Metal	56%	60%	65%	70 to 75%
Glass	32%	40%	43%	48 to 53%
Plastic - Overall	28%	30%	35%	40 to 45%

*most up-to-date recycling rates published by the different industry federations and PACSA

**estimations by the authors

In addition to the existing 2012 scenario, a fictional scenario for the municipality, in which LFG is not collected and used, was investigated. This was done because it is representative of most South African municipalities and reflects the case in the eThekwini Municipality before the CDM projects were implemented. Currently (2013) there are only two municipalities in the country which collect LFG on four of the landfill sites in the country. Of these landfill sites, only two flare it (Chloorkop and Robinson Deep in the Johannesburgh area) and two use it for electricity generation (Bisasar Road and Mariannhill in the eThekwini Municipality). Recycling was also included in this scenario in the same manner as explained in the previous paragraphs.

5.3.3.2 Scenario for 2014 and the closure of the Bisasar Road landfill site

The closure of the Bisasar Road landfill site and the re-directioning of the municipal solid waste to the other two existing landfills were investigated for the **year 2014** with regard to the GHG implications of these changes. The majority of the waste will be diverted to the Buffelsdraai landfill site and it was assumed that there will be no LFG collection at this landfill site in 2014 and 2015. A small percentage of the diverted waste will reach the Mariannhill landfill site, mainly due to private contractors collecting and transporting municipal solid waste. After consultations with personell at DSW it was assumed that 75% of the total municipal waste (excluding tyres, construction and demolition waste and 8% of garden waste) will be disposed at the Buffeldraai landfill site and 25% at the Mariannhill landfill site. There are plans that after the closure of the Bisasar Road landfill site about 8% of the 18% garden waste will be utilised at the closed site. In this scenario, GHG emissions from increased transport (40 km) were also included for the 68.5% waste that need extra transport from the city to Buffelsdraai, since these emissions would be additional to the current situation. Increasing recycling levels to the levels planned by PACSA as presented in Table 5.4 for the year 2014 have been included in calculations for emissions from recycling.

5.3.3.3 Scenario for 2020 and the improvement analysis of the changed waste system

Another series of calculations have been performed for the **year 2020** in a similar manner as for 2014. There are two variations of this scenario, namely the case in which LFG is not extracted at the Buffelsdraai LFS and the case in which LFG will be extracted and used for electricity generation by that stage. An improvement analysis with regard to GHG emissions for this year is also undertaken and it includes the increase in composting and recycling as well as the use of anaerobic digestion for the organic waste fraction (instead of composting).

5.4 Results and discussions

In a first stage, the amounts of each material ending up or expected to end up in landfills were calculated and are presented in Table 5.6 for each of the years modelled.

Table 5-6: Quantities of materials landfilled (tonnes)

Materials	**2012**	**2014**	**2020**
Organic waste	238091	253248	315407
Green waste	151865	161533	201181
LDPE	26408	28089	34983
HDPE	12234	13012	16206
PET	13837	14717	18330
Other mixed plastics	48344	51421	64043
White paper	13668	14538	18106
Mixed paper	31639	33653	41913
Cardboard	90782	96561	120261
Glass	60830	64703	80584
Metals	29192	31050	38671
Other	126807	134880	167986
Total	**843695**	**897407**	**1117671**

Base on the quantities shown in Table 5.6 and utilising the recycling rates for the respective years (see Table 5.5) the quantities of materials recycled were calculated and are presented in Table 5.7. In addition, the last row of this table presents the amounts of garden waste planned to be diverted.

Table 5-7: Recycled materials and composted garden waste for the years modelled (tonnes).

Materials	2012	2014	2020
Plastics – overall of which	43209	58463	90095
LDPE	11318	15125	23322
HDPE	5243	7007	10804
PET	5930	8644	13273
Other mixed plastics	20719	27688	42695
Paper – overall of which	195834	226407	349956
White paper	19668	22739	35147
Mixed paper	45529	52636	81360
Cardboard	130637	151031	233449
Glass	40554	48811	74385
Metals	43788	57665	90233
Total recyclables	323385	391346	604669
Garden waste for composting (8%)	0	71793	89414

By using the data from the above tables and the assumptions presented in the previous section, the GHG emissions from the management of waste in the eThekwini Municipality (excepting construction and demolition waste and tyres) have been calculated for the years 2012, 2014 and 2020 and are presented in Table 5.8 for the scenarios described previously.

Table 5-8: Tonnes of CO2 equivalents emitted or modelled to be emitted from the management of municipal solid waste in the eThekwini Municipality

GHG Emissions	2012		2014	2020	
	Without LFG collection*	With LFG collection		Without LFG collection at Buffelsdraai	With LFG collection at Buffelsdraai
Collection and transport of waste	9568	9568	10177	12674	12674
Landfilling with LFG collection and electricity production	0	-113967	-29825	-37146**	-148583
Landfilling without LFG collection	857447	55894	629304	783763***	0
Recycling (i.e. replacing virgin materials)	-113275	-113275	-334543	-519315	-519315
Composting	0	0	13282	16542	16542
Increased transport****	0	0	6275	7815	7815
Total	**753740**	**-161780**	**294670**	**264334**	**-630867**

*Representative for the majority of South African LFS
**Only from Mariannhill LFS
*** Only from Buffelsdraai LFS
****For the calculation of GHG emissions from increased transport an emission factor for trucks (0.190 kg CO_2 e/km/tonne) taken from literature (Eistedt et al., 2009) was used.

5.4.1 Overall results for 2012

The results for the year 2012 show that the overall GHG emissions from waste in the eThekwini municipality were negative, meaning that net GHG savings of -161 780 tonnes CO_2 e due to waste management in the municipality were calculated. These savings were mainly due to the collection of LFG and its use for electricity generation and due to recycling. The collection of LFG and subsequent electricity generation on the Bisasar and Mariannhill LFS saved -801 553 tonnes CO_2 e, representing the highest savings in the current system. The recycling of municipal solid waste saves 113 275 tonnes CO_2 by replacing virgin materials with recycled materials. Net emissions of GHG from the system were due to the collection and transport of municipal solid waste and due to the disposal of such waste in landfill sites with no gas collection and electricity generation (i.e. Buffelsdraai LFS). Compared to the savings, the net emissions for this year were much smaller.

For the 2012 scenario without LFG collection and electricity generation, which is representative of the majority of the landfills in South Africa, net GHG emissions of 753 740 tonnes CO_2 e were calculated. These would have been the emissions from the eThekwini solid waste management in the absence of the CDM projects. The large difference between the two scenarios for 2012 emphasises the importance of collecting the LFG and using it for electricity generation. In particular in South Africa, where electricity is produced primarily from coal, these savings are even larger compared to those possible in countries which do not have such a dependency on coal. Therefore, more South African landfill sites should be actively encouraged to invest in LFG collection systems and more substantial subsidies besides the current renewable energy ones (REFIT tariffs) should be made available in order to achieve substantial GHG savings from waste management.

5.4.2 Overall results for 2014

The calculations for 2014 show that the closing down of the Bisasar Road landfill site and the associated changes in the solid waste management system will cause, in the short term, an increase in GHGs due to solid waste management in the municipality. This is largely attributed to disposing larger quantities of waste in the Buffelsdraai landfill site which does not have LFG collection and to a smaller degree due to increased transportation. Although, in this scenario, the savings from recycling have more than doubled with the increase in recycling rates, these savings are not enough to counter the net emissions from the landfill disposal of waste.

Composting brings savings to this scenario. If a tonne of garden waste (excluding wood and with a degradable organic content of 20% (IPCC 2006)) is decomposing anaerobically in a typical South African landfill site, 66 kg of CH_4 are escaping resulting in 1 650 kg CO_2 e emissions. More details on the calculation procedures followed and the formulae used are presented in Friedrich and Trois (2013a). In these calculations it was assumed that 50% of the organic degradable carbon was decomposed (48% is stored and 2% is lost in leachate) giving rise to LFG. This LFG contains 55% CH_4 and 45% CO_2 (considered biogenic). If the storage of carbon from garden waste is included in calculations (i.e. the 48% degradable organic carbon from garden waste which does not decompose is locked away in the landfill site for 100 years and longer) and additional saving of -352 kg CO_2 e can be subtracted, this gives an overall net emission for 1 tonne of garden waste (no wood) of 1 298 kg CO_2 e. If the same amount is composted the emissions per tonne are of about 185 kg CO_2 e for turned windrow composting and 172 kg CO_2 e for aerated dome composting, as practiced in South Africa (Friedrich and Trois, 2013b). For both composting technologies, these are important savings of 1 113 and 1

126 kg CO_2 e, respectively, per tonne of garden waste. This means that although composting in this context has net GHG emissions, the savings from composting, as opposed to landfilling garden waste are important. If in 2014 about 71 793 tonnes of garden waste will be composted through turned windrows (the technology with the highest emissions), the GHG emissions due to this activity will amount to about 13 282 tonnes CO_2 e. In the absence of composting these emissions would be 93 187 tonnes CO_2 e. Therefore, through composting alone the GHG emission from the normal collection and transport of municipal solid waste, as well as from increased transport due to the closure of the Bisasar Road landfill, can easily be offset.

5.4.3 Overall results for 2020

The overall results for 2020 show that the GHG emissions from the eThekwini Municipality solid waste system can amount to either savings or increased net emissions, depending on the implementation of a LFG collection system with electricity generation. If no such system is implemented, then the emissions for 2020 are much higher than those from the current scenario but slightly less than those from the year 2014, although the amount of municipal solid waste increased in this period. This is due to increased recycling and composting. This shows that increased recycling and composting has the potential to stabilise GHG emissions due to increase in waste generation but it is not enough to make the waste management carbon neutral or to lead to GHG savings. GHG savings are possible only with LFG collection and utilisation. If a LFG collection system with electricity generation is implemented, net GHG savings will be achieved, giving the best outcome for the overall waste management system. Such systems are expensive and require investments, and therefore the CDM mechanism was very important for the South African local authorities to establish the first few LFG collection and utilisation projects to show that high GHG savings are achievable in the local context. In the next section a series of possible improvements are explored for the waste management system for the year 2020.

5.5 Additional GHG savings possible for 2020

Three possible interventions considered feasible locally for achieving additional GHG savings from the management of municipal solid waste during 2020 were investigated, namely increased recycling, garden waste composting and the anaerobic digestion of organic waste fraction available (food waste and garden waste). The results from this modelling exercise are summarised in Table 5.9 and they are compared with the results already modelled for this year and presented in Table 5.8. In calculating GHG emissions for increased recycling (paper) and composting, as well as for anaerobic digestion, the carbon diverted from landfills will not be available for anaerobic decomposition and will not produce LFG. This is illustrated in Table 5.9 (see row entitled landfilling without LFG collection).

Table 5-9: Different GHG emissions from waste management in the eThekwini Municipality in 2020 (tonnes CO_2 e)

GHG Emissions	2020				
	Without LFG collection at Buffelsdraai*	**With LFG collection at Buffelsdraai**	**With increased recycling**	**With increased composting**	**With anaerobic digestion**
Collection and transport of waste	20489	20489	20489	20489	20489
Landfilling with LFG collection and electricity production	-37146**	-148583	-37146**	-37146**	-37146**
Landfilling without LFG collection	783763***	0	447335***	580660***	448494***
Recycling (i.e. replacing virgin materials)	-519315	-519315	-654276	-519315	-519315
Composting	16542	16542	16542	28948	0
Anaerobic digestion	0	0	0	0	-70204
Total	**264334**	**-630867**	**-207056**	**73636**	**-157682**

*Representative for the majority of South African LFS

**Only from Mariannhill LFS

*** Only for Buffelsdraai LFS

Increased recycling due to an improved overall recycling rate of 10 % (as opposed to 5% used to produce the figures in Table 5.8) will lead to savings amounting to -654 276 tonnes CO_2 e. Therefore, a change in the overall recycling rate of 5% will increase the GHG savings due to this activity by about 20%, showing that the recycling rate is a sensitive parameter in this case. Such high recycling rates are achievable in South Africa with concerted efforts from all the players involved in recycling in the country. For example India, another developing country has much higher recycling rates (Chintan, 2009) than those projected for 2020 for South Africa. For example, in Dehli, recycling rates of 95% are reported for mixed paper and 70% for metals and mixed plastics (Chintan, 2009). The increase in the rate of paper recycling and aluminium will have the highest impacts for reducing GHGs.

In addition to replacing virgin paper, the recycling of paper reduces the degradable organic content of the municipal solid waste, leading to decreased emissions from the landfilling of waste in the municipality. Since paper has a high degradable organic content (40% for paper in wet waste – IPCC, 2006) it makes a difference in reducing landfilling GHG emissions. This together with the savings due to replacement of virgin paper, will result in net savings from waste management (-207 056 tonnes CO_2 e) for the year 2020.

Increased composting can bring about another set of potential savings in the municipal solid waste management system. If the garden waste composting increases from 8% to 14% of the total waste (which means that about 78% or 156 474 tonnes of the available garden waste should be collected separately and used for composting) the GHG emissions are calculated to be 28 948 tonnes. If the same amount of garden waste would have been disposed in a landfill site without gas collection the GHG emissions due to this waste would amount to 203 103 tonnes CO_2 e. If the same amount would be disposed in a landfill site with LFG collection and electricity generation the GHG emissions would be negative (i.e. -42 886 tonnes CO_2 e) amounting to net savings. Diverting garden waste from the municipal stream towards composting also leads to a decrease in the organic content of the solid waste landfilled. However, because the degradable organic content for garden waste is on average 20% (IPCC, 2006), these savings are much lower per tonne of garden waste diverted as compared to paper waste, but higher when compared to food waste, which has an average of 15% degradable organic carbon. Also in this scenario the amounts diverted are moderate and as a result the overall GHG reductions due to this diverting process are not that high and the overall system waste management system has net GHG emissions of 76 636 tonnes CO_2 e.

Anaerobic digestion of organic waste consisting of food waste and garden waste is another avenue that could lead to lower GHG emissions. An emission factor of -0.2718 tonnes CO_2 per tonne of wet biogenic waste was calculated for the local context by Jagath and Trois (2011) and is used in this study. This factor includes the use of the resultant biogas for the generation of electricity and the digested matter as fertiliser. Assuming that half of the biogenic waste in the eThekwini Municipality, comprising of green waste (from gardens and parks) and other organic waste (e.g. food waste), will be collected and diverted to anaerobic digestion, for the year 2020 this represents about 258 294 tonnes. The anaerobic digestion process with electricity production will result in savings of -70 204 tonnes of CO_2 e. If the same amount of this type of waste would have been disposed in a landfill site without LFG collection the net emissions from this waste would amount to 335 265 tonnes CO_2 e. If the same amount would be disposed in a landfill site with LFG collection and electricity generation the savings would be -70 793 tonnes CO_2 e. When considering the entire system the introduction of anaerobic digestion has the potential to save -157 682 tonnes CO_2 e due to the diversion of organic degradable carbon from landfill sites and due to the generation of electricity and fertiliser.

Anaerobic digestion, recycling of paper and composting of garden waste are in direct competition for carbon with landfilling. Composting is also competing for inputs with anaerobic digestion. It is possible to maximise GHG savings by taking into account these competing processes and combining different waste management strategies in an integrated waste system as to achieve higher savings. Increasing recycling of different materials, but in particular of paper, can work very well together with anaerobic digestion, since there is no competition for waste materials. In fact, this combination would achieve higher GHG savings than recycling combined with composting, but lower than LFG collection and utilisation systems.

Calculations done for this paper show that in the absence of a LFG collection system and the generation of electricity at the landfills, anaerobic digestion, recycling of paper and composting are better options with regard to GHG emissions. However, in the presence of a LFG collection system and with electricity generation, the landfilling of paper and biogenic waste achieve the highest GHG savings. Recycling and composting have other advantages in terms of lower investment needed and the potential to create jobs locally. These are important factors to consider when planning an integrated waste management system for the future of the municipality. Therefore, the implementation of a LFG collection system for the Buffelsdraai landfill site in the future will have serious implication on the decisions regarding anaerobic digestion of biogenic waste, composting of garden waste and recycling of paper in the municipality. These actions should be considered together, since the paper and biogenic waste will not be available for methane and electricity generation if recycled, composted or anaerobically digested. Therefore, this study can aid decision making with regard to the GHG implications of the scenarios possible.

5.6 A brief overview of uncertainties and limitations

There are two main types of uncertainties affecting the results presented in this paper. The first set is linked to the use of a methodology which is based on emission factors and calculates instantaneous GHG emissions as opposed to time related GHG emissions. This affects GHG emission factors from landfills and dumps, since the decomposition of the carbon in waste happens over years. Other GHG emission factors (e.g. for collection and transport and recycling) do not have this time factor associated as these activities happen all the time but are short lived. The second set of uncertainties is linked to the parameters on which the GHG emission factors are based and the future projections of these parameters.

When landfill GHG emissions are calculated by using emission factors (which are not time dependent) then a methane commitment approach is used. The comparison of this approach to the waste in place method (which considers time in the decomposition of waste), showed methane commitment to be more suitable for planning applications and less suitable for GHG inventorying (Mohareb et al., 2011). Other points of criticism have been presented by the same authors and are linked with the uncertainties in the calculation of carbon savings as well as with the fact that the inclusion of these savings "*might shift the focus away from current methane emissions*" (Mohareb et al., 2011). For the current research, this criticism is only partially valid for the calculated results presented. By comparing the different scenarios as presented in Table 5.8 and Table 5.9 it is obvious that methane emissions from landfill gas are very important and if not collected and utilised they dominate the GHG profile of those scenarios. Therefore, the inclusion of recycling and other processes saving GHG does not shift away the focus from methane emissions. In fact, it highlights the importance of methane collection and utilisation. The inclusion of GHG savings due to recycling, replacement of fossil fuel based energy and sequestration of biogenic carbon in landfill sites, might introduce a margin of error when calculating GHG emissions, but leaving GHG savings out will present even higher uncertainties and an incomplete system. The margin of error in the results due to the inclusion of these savings has been reduced, as much as currently possible, by using local data in the calculations and by developing local GHG emission factors.

The second set of uncertainties linked to the parameters on which the GHG emission factors are based and their future projections influence the way GHG emission factors are calculated. GHG emissions factors calculated for South Africa will change if the parameters used for their calculation change. For GHG emission factors from landfills such parameters are the precise composition of the waste, the methane oxidation factor, the landfill gas collection factor and the flaring/energy efficiency factor. Preliminary work (Friedrich and Trois, 2013a) investigating these parameters showed that the methane oxidation factor had the highest significance with regard to the calculation of local emission factors for landfill sites. However, no data exist on methane oxidation for South African landfill sites and a conservative approach based on published literature data was followed with methane oxidation. Therefore, for landfill sites without LFG collection in particular, the local GHG emission factors might be higher than what is happening in reality. This might be also influencing the scenarios presented in Table 5.8 and Table 5.9 in this study. Therefore, local investigations of methane oxidation in South African landfills are needed to further improve the accuracy of calculations.

The calculations for the future scenarios were based on the assumption that waste generation trends will continue in the future as in the past. For the eThekwini Municipality this translated into a constant increase in the waste generated and disposed in landfills. However, not all South African municipalities show this trend. For example, in the City of Cape Town declines in waste generated (total and per capita) have been recorded in recent years (City of Cape Town, 2011). Socio-economic factors determine these trends and more research is needed into how these factors influence the amounts of waste generated in the different municipalities in the country. If such declining trends will be experienced in the eThekwini Municipality the calculations for this study need to be adjusted. Overall, lower amounts of waste lead to lower GHG emissions and the extension of the lifespan of existing landfills. For the Bisasar Road and Mariannhill landfill sites this has additional benefits since the waste would be landfilled in landfills with gas extraction and electricity generation. For the Buffelsdraai landfill site the pressures to install landfill gas collection and utilisation and the related expenses can be postponed for a few years which will give the municipality more time to raise the necessary funds.

5.7 Conclusion

By taking into account a life cycle perspective of the municipal solid waste system of the eThekwini Municipality it was calculated that for the year 2012 net savings of -161 780 tonnes CO_2 e were achieved. This is mainly due to the landfill gas to electricity CDM projects and due to recycling in the municipality. The landfill gas to electricity project is critical for the current and future GHG emissions from the local management of municipal solid waste. In the absence of LFG collection and utilisation systems, which is typical for the majority of South African landfills, important GHG emissions from the anaerobic degradation of waste are recorded.

The closure of one of the three local landfill sites and the re-directioning of the majority of waste to another landfill site, which does not have LFG collection and utilisation, will cause an increase of GHG emissions to 294 670 tonnes CO_2 e. In the future, an increase in recycling and the introduction of anaerobic digestion and composting has the potential to reduce these emissions. However, only the introduction of a LFG collection and utilisation system will result in the highest possible overall GHG saving from waste management in the municipality. In the absence of the CDM mechanism, the LFG collection and utilisation system has to be financed locally and might present a financial challenge to the municipality. Therefore, the second intervention which will make a difference by lowering GHG emissions from waste management would be to increase recycling in general and in particular the recycling of paper. Since there is no direct competition for carbon, in addition to recycling, anaerobic digestion can be introduced and this combination will achieve increased savings in the future. If anaerobic digestion is not possible, composting in addition to recycling will also lead to savings, albeit not that high as with anaerobic digestion.

The results presented in this paper show that life cycle based GHG emission factors for waste and their use can support decision-making for municipalities in the local context. They can give valuable input for the planning and development of future waste management strategies and they can help optimising municipal solid waste management. As more research is done locally these emission factors can be further improved. In particular methane oxidation in landfills has to be researched in more detail in the local context.

In addition, to the scenarios and possibilities presented in this paper, it would be interesting to study further aspects not included. For example, the GHG implications of incinerating municipal waste with energy recovery have not been studied since there is historical opposition to incineration in the country. Furthermore, other more recent disposal technologies like pyrolysis also need more investigation in the local context. Other probable waste management technologies as well as other possible combinations of different waste management strategies can also be further researched.

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Chapter 6 Conclusion and Recommendation

6.1 Introduction

This chapter summarises the findings of this research and presents recommendations for future work in the field. Sustainable development is the overarching concept behind this dissertation, with global climate change as one of the most important environmental problems currently facing mankind. It is caused by the anthropogenic release of GHGs and the waste sector is also a contributor, albeit minor. However, this sector has the potential to become a major saver of GHGs but for municipalities in developing countries and in particular for African municipalities, methodologies to account, report and optimise waste management processes are lacking. This study addressed this research gap in the South African context, focusing on one local municipality – the eThekwini Municipality.

6.2 The main findings and conclusions of the study

The literature review for this study presented in Chapter 2 compared the GHG emissions from different municipal waste management processes in developing and developed countries, with particular emphasis on the African continent and South Africa. What sets developing countries apart are the different motivational factors for GHG accounting and reporting. Developing countries do not have a mandatory obligation to report GHG and there are less data and information for waste management in general and in particular for the quantification of greenhouse gases. In the absence of such data, a variety of assumptions have to be used, which affects the accuracy of calculations and makes validation of results a very challenging process. One example of such an assumption is the waste generation rate for African countries (IPCC, 2006) which currently seems to be over-estimated.

When investigating theoretical GHG emissions from individual waste management processes there is agreement on the magnitude of the emissions expected from each process (generation of waste, collection and transport, disposal and recycling). Theoretically, recycling brings about the highest savings in terms of GHG, followed by composting and incineration with energy recovery. The disposal of waste in landfills has some of the highest GHG emissions. In particular, in developing countries these emissions are dominating due to the methane released by dumpsites and landfills. If these are upgraded to sanitary landfills, these emissions will continue to increase, unless the methane is captured and either flared or used for electricity generation. The CDM projects have made some in-roads with regard to the waste to energy projects, however, the African continent lags behind. The GHG emissions from transport and collection are lower in developing countries due to inadequate provision of these services, in particular in African cities which have some of the lowest collection rates.

The results and analysis presented in Chapter 3, show that in the accounting and reporting of greenhouse gases from waste in South Africa, the most important role is played by calculated inventories, followed in recent years by the use of CDM procedures which allow direct measurements of greenhouse gases (mainly methane) from landfill sites. Although not bound by the Kyoto Protocol and with no national legislation forcing disclosure, South Africa has produced three national inventories (for the years 1990, 1994 and 2000) and most of the largest municipalities in the country have produced municipal greenhouse gas inventories too. One of the most important limitations of these inventories is the change in the methodology used for calculating greenhouse gases from landfills – i.e. the replacement of the gas yield model (also called methane commitment model) by the first order of decay model. This change is in line with IPCC guidelines and should achieve more realistic results and higher overall accuracy; however, the calculated emissions of successive inventories produced by the different methodologies should not be compared. For South Africa the results for the 1990 and 1994 inventories cannot be compared with those for 2000, as they show a decrease of greenhouse gases due to waste which is incorrect. In reality the emissions have increased substantially. Establishing a trend requires recalculations by using the same methodology. However, these interpretations should be specified in the national inventory and particular in summary tables and not left open to misinterpretation.

When it comes to municipal greenhouse gas inventories from waste, the methodological choice is even wider and five different ways for greenhouse gas accounting and reporting are presented in the literature. In South Africa, the methodology used was influenced by the affiliation to international initiatives, which promoted the production of the municipal inventories. The first municipal inventories (which included emissions from waste) were produced at the beginning of the 2000s under the initiative of the CCP (Cities for Climate

Protection) campaign launched through the ICLEI (International Council for Local Environmental Initiatives). For the eThekwini Municipality there were three successive inventories (for the years 2001-2002, 2005-2006 and 2010-2011) prepared and a fourth one is in its initial stages. For the waste sector, some conclusions are pertinent, namely: the quality of the inventories produced has increased due to the interaction with the CDM landfill gas to energy projects and due to the development of local capacity to undertake municipal greenhouse gas inventories. Therefore, CDM projects have the indirect benefit of increasing the quality of local greenhouse gas inventories due to waste and setting a standard for how greenhouse gases should be accounted and monitored at landfill sites. However, accounting of GHG emissions from municipal waste only by using inventories has shortcomings. The most important of these is that many processes which are part of the waste management system of a municipality (e.g. collection and transport, as well as recycling) are not included, therefore, holistic planning and the calculations of trade-offs between different waste strategies available is not possible. To address these shortcomings, life cycle based emission factors had to be derived for the different waste management processes used in South Africa.

The development of emission factors for waste management processes is important for municipalities in developing countries, because many of these municipalities do not have all the data needed for a thorough GHG accounting process. However, few developing countries have attempted developing emission factors to suit specific national conditions with the result that almost all published factors emanate from Europe and North America. The research undertaken and presented in Chapter 4 is intended to fill this gap in the context of South Africa. For collection and transport of waste, the average diesel consumption is about 5 dm^3 (litres) per tonne of wet waste collected and transported in the country and the average GHG emissions are about 15 kg CO_2 e. These averages are in the range published in the international literature for lower density urban areas. However, for the collection and transport of municipal waste in the rural fringes of municipalities, South African GHG emissions are much lower than the international ones. This difference is due to the lower collection rate in these locations and not due to higher efficiencies in the collection processes.

The GHG emission factors developed for landfill sites show that the highest contributions per unit of wet waste in this country are due to landfill sites without gas collection. These are the majority of the landfill sites in the country and with the upgrading of dumps, more and more of the local municipal waste will be disposed in this way. This situation is valid for the majority of municipalities in the developing world and will lead to negative consequences for the emission of GHG. Therefore, more efforts should be made to reduce these emissions by designing covers to maximise methane oxidation and by collecting and utilising LFG. However, lack of financial resources is limiting the implementation of LFG collection systems in South Africa. Possible financial avenues to increase the implementation of such systems are the recently introduced renewable energy tariffs and the CDM projects. The CDM projects, which are instrumental for LFG collection and electricity production in the country proved very successful, and these landfill sites have the lowest GHG emissions. However, future extension of these projects to other landfill sites is uncertain. Currently there is no other mechanism to replace the Kyoto agreements and the subsequent CDMs, which provide an external source of funding for LFG collection. For cash-strapped municipalities in the developing world this is a negative outcome.

Another important conclusion from the development of GHG emission factors for landfill sites is that inclusion or exclusion of carbon storage is important for calculations. More studies should be undertaken in order to develop a common approach to future GHG emission calculations. In the meantime, GHG reporting from landfill sites should give enough details on the calculation procedures and how emission factors have been aggregated.

The results for recycling of municipal solid waste show that all recycling processes bring about net GHG savings. These savings range from -290 kg CO_2 e (glass) to – 19 111 kg CO_2 e (metals - Al) per tonne of recyclable. Most of the GHG factors calculated for South Africa are in the ranges presented in the literature, with the exception of energy intensive materials (i.e. aluminium and steel) where the GHG savings due to recycling are higher in South Africa. These results underline the interdependency of the waste management system of a country/region with other systems, most importantly the energy system, with South Africa having a carbon intensive energy generation system dependent on coal.

Composting of garden waste, as it was done in the few studies undertaken at UKZN in Durban, has net GHG emissions (about 185 kg CO_2 e per tonne of wet garden waste for turned windrow and 172 kg CO_2 e per tonne of wet garden waste for the aerated dome technology). These results are also within the ranges presented in the literature. However, more research is needed to refine these figures for more arid parts of the country which have large municipal areas (e.g. Johannesburg) and which are important waste generators.

Chapter 5 presents the current GHG emissions from municipal solid waste and projected GHG emissions for the years 2014 and 2020 for the eThekwini Municipality. These figures have been calculated by using the emission factors developed and presented in Chapter 4. It was calculated that for the eThekwini Municipality for the year 2012 net savings of -161 780 tonnes CO_2 e were achieved. This is mainly due to the landfill gas to electricity CDM project and due to recycling in the municipality. The landfill gas to electricity project is critical for the current and future GHG emissions from the local management of municipal solid waste. In the absence of LFG collection and utilisation systems, which is typical for the majority of South African landfills, important GHG emission from the anaerobic degradation of waste are recorded.

The closure in 2014 of one of the three local landfill sites and the re-directioning of the majority of waste to another landfill sites which does not have LFG collection and utilisation, will cause an increase of GHG emissions to 294 670 tonnes CO_2 e. In the future (2020) an increase in recycling and the introduction of anaerobic digestion and composting has the potential to reduce these emissions. However, only the introduction of a LFG collection and utilisation system will result in the highest possible overall GHG saving from waste management in the municipality. In the absence of the CDM mechanism, the LFG collection and utilisation system has to be financed locally and might present a financial challenge to the municipality. Therefore, the second intervention which will make a difference by lowering GHG emissions from waste management would be to increase recycling. Since there is no direct competition for carbon, in addition to recycling, anaerobic digestion of food and garden waste can be introduced and this combination can achieve increased savings. If anaerobic digestion is not possible, composting in addition to recycling will also lead to savings, albeit not as high as with anaerobic digestion.

The results presented in this study show that life cycle based GHG emission factors for waste and their use can support decision-making for municipalities in the local context. They can give valuable input for the planning and development of future waste management strategies and they can help optimising municipal solid waste management. As more research is done locally these emission factors can be further improved.

6.3 Research question and objectives revisited

The research question of this study was:

What are the best local waste disposal practices/strategies that will ensure an effective reduction of greenhouse gas emissions from the management of municipal solid waste?

The question has been answered in Chapter 5 and the results presented show that LFG collection and electricity generation will achieve the highest GHG savings from the management of municipal waste. Other individual disposal practices like recycling, anaerobic digestion and composting can also achieve savings but if these practices are used in isolation the savings are not that high. To maximise these savings an integrated approach in which LFG collection and electricity generation is complemented by increased recycling should be used. Anaerobic digestion combined with increased recycling can achieve higher savings than composting combined with increased recycling. In this integration and optimisation process there is competition for the carbon in the waste, which has to be considered.

The objectives of the study were as follows:

1. Review existing information on the quantification of GHG from the management of municipal solid waste, emphasising the situation in developing countries and in particular in Africa and South Africa.

This objective has been achieved in Chapter 2 and a comparative literature review is presented.

2. Review current GHG accounting and reporting from the waste sector at local and national level.

This objective has been achieved in Chapter 3 and the results showed that current accounting and reporting relies on the use of GHG inventories. However, national and local inventories have shortcomings and GHG emission factors for individual waste management processes for South African municipalities were needed.

3. Develop South African GHG emission factors for the different waste management processes used in the country.

This objective has been achieved in Chapter 4 and local GHG emission factors have been developed for the collection and transport of waste, for disposal of waste in landfills and dumps, for recycling and for composting.

4. Use the above factors to calculate current and future GHG emissions from waste management for the eThekwini Municipality and to investigate GHG reduction opportunities

This objective has been achieved in Chapter 5 and current GHG emissions, as well as the future projection from the management of municipal solid waste in the eThekwini Municipality are presented. It has been shown that an integrated approach which uses more than one waste management strategy will achieve the highest savings in terms of GHG reductions.

6.4 Recommendations for further research

The GHG emissions from waste management in developing countries are predicted to increase exponentially. Therefore, more attention has to be paid to how these emissions arise, are accounted, calculated and reported for waste management processes in the municipalities of developing countries. One of the major problems for these municipalities is the lack of data on the municipal waste itself, as well as for the waste management processes used. In South Africa, during the course of this research, it was observed that small municipalities usually lack the financial, technical and managerial capacity to collect waste data. This observation is also supported by literature. Therefore, the local need for more reliable data for waste generation and waste composition to calculate GHG emissions, also adds to the initiative to develop a national waste quantification and information system. Although there are challenges with regards to the implementation of such a system in all South African municipalities, the final result, even with partial implementation, will lead to a better input for further research on GHG emissions from waste. As a result, an important recommendation is to support in every way possible the national waste quantification and information system and to speed up its implementation. Further studies should improve the quality of waste data in large municipalities (e.g. more frequent waste stream analyses), but should also gather waste data for smaller municipalities which have no waste data at all. Innovative approaches like, for example, compulsory vacation work or internships involving senior students for these kinds of data gathering projects, or the use of community work for professional registration for engineers in this areas, need to be investigated.

A direct comparison of GHG emissions from waste management in different municipalities should be undertaken only at process level (i.e. collection and transportation of waste, landfilling, recycling, composting, etc.). At systems level (entire municipalities or large parts of them) such comparisons should be avoided, because the determining factors (e.g. waste composition, collection rates, waste management process, carbon storage, etc.) are different and so could be the accounting methodology used (i.e. methane commitment approach vs. waste in place approach). Therefore, there is a need to develop a common approach applicable for the accounting of greenhouse gases from waste management at municipal level and individual processes should be the foundation blocks. As a result, further studies should perfect a methodology which can be applied by South African municipalities and an accounting and comparison tool (like the WARM spreadsheet in the United States) should be developed. This tool will aid municipal managers in charge of waste management and will lead to better decision-making in this context. The development of local GHG emission factors is a first step in this direction.

The emission factors presented in Chapter 4 can be improved by conducting more local research. In the South African context, where significant amounts of waste are still disposed in landfills and dumps, more data and research is needed on the dumps themselves, as well as on the overall performance of landfill sites in the presence and in the absence of LFG collection systems. In particular, methane oxidation in landfill covers for landfills without gas collection systems needs investigation, as it has the potential to significantly reduce GHG emissions from these sites without needing high investments. Therefore, future research is needed to establish real methane oxidation values for the local landfill sites. Also, more research is needed for the improvement of the local GHG emission factor for plastic recycling, because some European data had to be used in the calculations. The possible regional variations of the local GHG emission factor for composting also needs more research, since it was deduced from data from a single location and there are large differences within South Africa with regard to average rainfall and temperatures.

In addition, to the scenarios and possibilities presented in this study, it would be interesting to investigate further aspects not included. For example, the GHG implications of incinerating municipal waste with energy recovery have not been studied, since there is historical opposition to incineration in the country. Furthermore, other more recent disposal technologies like pyrolysis also need more investigation in the local context. Other probable waste management technologies and strategies, as well as other possible combinations of these, should also be further researched.

The influence of socio-economic factors on the generation of waste in the different municipalities in the country should also be investigated, since it is of importance for the prediction of the amounts of waste that need disposal and contribute to GHG emissions. Waste generation scenarios informed by socio-economic factors could be used for further modelling in order to increase the realism of GHG emissions predictions.