



Landfills gas emissions and the associated air quality, energy and climate change implications in South Africa

Shaazia Bhailal

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Declaration

I declare that this thesis is my own, unaided work, except where otherwise acknowledged. It is being submitted for Degree of Doctor of Philosophy at the University of the Witwatersrand, Johannesburg. It has not been submitted before for any degree or examination in any other university.

Signed this ____ day of _____ 2016

Shaazia Bhailall

Abstract

Global methane (CH_4) emissions are divided mostly into three sources; biogenic, thermogenic and pyrogenic. The sources can be anthropogenic or natural in origin. Anthropogenic sources include emissions associated with agriculture (rice paddies and ruminants), waste (landfill and waste water), biomass burning and fossil fuels. Landfills have been implicated as one of the largest anthropogenic sources of atmospheric CH_4 globally and as a significant contributor to global warming. According to the Intergovernmental Panel on Climate Change (IPCC) Fifth Assessment Report anthropogenic sources account for 304 – 368 $\text{TgCH}_4/\text{year}$ and methanogenesis in landfills and waste contributes between 67 and 90 $\text{TgCH}_4/\text{year}$ to this amount (between 22 and 24% of emissions).

Although CH_4 and carbon dioxide (CO_2) from landfills are produced in about equal amounts, CH_4 is of greater concern as a greenhouse gas. The global warming potential of CH_4 is approximately 28 times greater than that of CO_2 over a 100 year period. In addition CH_4 has a shorter atmospheric lifetime when compared to CO_2 . Therefore, quantifying CH_4 emissions from landfills is essential from an environmental, regulatory and financial perspective as well as to evaluate measures for reduction of national and global greenhouse gas emissions. The main aim of this study was to estimate landfill gas (LFG) emissions in South Africa and individual landfills using theoretical models (LandGEM, GasSim, IPCC Waste Model 2006 and the California Landfill Methane Inventory Model (CALMIM)). The model

estimations are compared to actual gas yields from LFG extraction and utilization projects at individual landfill sites to assess the applicability of the models to South African sites. Direct surface emissions fluxes were also measured and compared to model estimates. The polluting potential of landfill sites was also assessed using the Robinson Deep landfill site in Johannesburg as a case study. Modelled pollutant emissions using AERMOD are compared to ambient emissions that was observed at the landfill site using a mobile air quality monitoring caravan. LFG extraction and utilisation projects implemented in South Africa are also assessed. Six projects (implemented at 10 landfill sites) in South Africa i.e. Ekurhuleni Metro Municipality, City of Johannesburg, two within eThekweni Metro Municipality, Chloorkop and Alton landfill sites were evaluated to gauge the applicability and success of each project under the Clean Development Mechanism (CDM) programme of the Kyoto Protocol.

IPCC Waste Model estimates show that Gauteng is the largest contributor (48.36%) of CH₄ emissions from waste disposal in South Africa followed by the Western Cape. City of Johannesburg is the main contributor amongst the major Metro Municipalities in the country. In total in South Africa 531 735 tCH₄/annum and 635 207 tCH₄/annum emission were predicted for 2015 and 2020 respectively. The results, showing that Metro Municipalities make a significant contribution to emissions, are indicative of the vast disparities in waste disposal between urban and non-urban areas in the country. Comparing actual gas generation at individual landfill sites to LFG generation estimates using LandGEM, GasSim and IPCC Waste Model 2006 show that LandGEM and IPCC Waste Model are very similar and are shown to over predict LFG generation. GasSim simulates much lower

emissions and falls within actual gas yields at some sites. Thus, using the IPCC Waste Model to estimate the contribution from the solid waste sector to national CH₄ emissions in South Africa could be problematic and result in an over estimation of emissions. This also highlights the importance of good quality data used as input into models to estimate LFG generation. In South Africa, only a few landfill sites were operational since 1950 with very minimal to non-existent data being available on the operation of the sites. More often than not, waste characterisation information is not available as well as waste tonnages collected at sites (as many landfills in South Africa do not have operational weighbridges). In additions the models do not take into consideration site specific characteristics like; material used and thickness of cover soils, the impact of climatic conditions on emissions, the seasonal variability of CH₄ oxidation rates in cover soils as well as daily operations at a specific site (for example whether or not waste is compacted sufficiently and covered on a daily basis). This information is vital for estimating LFG generation.

Gas fluxes measured at the Robinson Deep and Marie Louise landfill sites using static flux chambers show that CH₄ and CO₂ emissions from the surface of the landfills are fairly low. CH₄ fluxes range between -0.0024 g/m²/day and 0.17 g/m²/day at the Robinson Deep landfill site. The mean CH₄ flux at Marie Louise was 2.04 g/m²/day and ranged between 0.0005 – 0.44 g/m²/day. Flux measurements taken at a closed section where an LFG extraction system was installed at Robinson Deep was 0.01 g/m²/day, reflective of the LFG extraction system aiding in decreasing emissions to the surface. CO₂ emissions were higher than CH₄ emissions at both sites with fluxes ranging between -2.87 – 13.27 g/m²/day and 0.60 – 2.37 g/m²/day at Robinson Deep and

Marie Louise landfill sites respectively. The higher CO₂ emissions relative to CH₄ emissions could be attributed to CH₄ oxidation in cover soils, however this was not determined in this study. Emissions estimations from daily cover soils using CALMIM are 16 times higher compared to measured emissions from the surface and emission estimates from intermediate and final cover soils are much higher. The low CH₄ and CO₂ surface fluxes could be attributed to the slow degradation of waste at the surface of the active cell. The waste degrades aerobically and reaches methanogenesis at a later stage or once another layer of waste is disposed above. High oxidation rates of CH₄ in cover soils to CO₂ also result in lower emissions to the atmosphere. This highlights the use of theoretical models in estimating emissions without the consideration of site specific characteristics such as cover material, climatic conditions and site specific CH₄ oxidation rates in cover soils. This is critical when determining the contribution of landfill emissions to national greenhouse gas inventories. The current IPCC Waste Model used in South Africa to determine national emissions from the waste sector over estimates the contribution of landfill emissions, sometimes by as much as 60%, when compared to actual gas extraction rates at individual landfill sites.

Despite the low measured CH₄ and CO₂ emissions from the landfill surface, methanogenesis is reached within the landfill site as shown by the LFG extraction rates. Of the ten landfill sites in South Africa that have implemented LFG extraction and utilisation projects only eight are still operational and two are successfully generating electricity. Despite the many technical and administrative challenges faced, the projects have had many sustainable development benefits for South Africa as well as mitigating close to 3 million tCO₂e. Capital costs, daily

operational and maintenance expenditure as well as costs associated with the registration of projects, verification and transactional advisors for the sale of emission reductions currently make CDM projects financially unviable. CDM projects could become financially profitable with the potential implementation of a carbon offsets programme in South Africa as part of the Carbon Tax bill. This will allow CDM projects to trade their emission reductions at a much higher rate than what is currently available on the global carbon market.

This thesis is dedicated to:

Joe, KB and Siyabonga

For days and nights spent with me collecting data at landfill sites. I am forever grateful

My parents

For providing me with all the support and opportunities throughout my life

My son

For being my strength, inspiration and motivator

My husband

For everything

"Whatever is in the heavens and the earth declares the glory of God, and He is the Mighty, the Wise"

Quran 57:1

"Whoever is granted wisdom has indeed been granted wealth abundant"

Quran 2:269

Preface

The principal gases including CH₄ and CO₂ from landfills are produced by the decomposition of the organic fraction of waste. Landfill CH₄ emissions are a significant source of anthropogenic CH₄ emissions and play an important role with regard to the greenhouse effect. In addition, landfill gas (LFG) emissions are highly variable both temporally and spatially (Scharff and Jacobs, 2006). Quantification of LFG emissions is important to evaluate measures for greenhouse gas emission reductions and to contribute to national greenhouse gas inventories (IPCC, 2006). Several models to estimate gas emissions from landfills have been developed mainly for regulatory purposes. Most models are based on first-order kinetics (FOD) using single phase or multiphase equations to simulate emissions (Scharff and Jacobs, 2006). FOD models were mainly used to quantify the potential of LFG production from a landfill to assess the applicability of LFG extraction at a specific site. Most models do not consider site specific characteristics like; material used and thickness of cover soils, climatic conditions and seasonal variability of CH₄ oxidation rates in cover soils (Bogner *et al.*, 2014).

The principal aim of this study is to evaluate LFG generation and emission rates from landfills in South Africa by analysing subsurface LFG data and estimating emissions using theoretical models. The polluting potential and its contribution to pollutants in ambient air is assessed. The success of LFG extraction and utilisation projects under CDM is also assessed.

The main objectives of this study are to:

1. determine the LFG production and emissions in South Africa using the gas estimation models LandGEM, GasSim, IPCC Waste Model 2006 and the California Landfill Methane Inventory Model (CALMIM).
2. compare predicted gas generation to actual gas yields from LFG extraction systems and measured CH₄ and CO₂ fluxes from the surface of the landfill site.
3. determine the air quality impact potential of landfill sites.
4. determine the success of landfill gas extraction and utilisation projects in South Africa under CDM of the Kyoto protocol.

This dissertation is divided into six chapters. Chapter 1 provides the necessary background information relating to waste degradation processes and resultant LFG production and emissions as well as waste management legislation in South Africa. Chapter 2 provides a description of data collection and analysis methodologies. In Chapter 3 monitored LFG production rates at the landfills are analysed together with LFG production simulated by the LandGEM, GasSim, IPCC Waste Model 2006 estimation models. Flux measurements from the two landfill sites are also presented and compared to emissions estimates using CALMIM. The polluting potential of the Robinson Deep landfill site is presented in Chapter 4 using ADMS and compared to monitored ambient pollutant concentrations. Chapter 5 analysis the success of LFG extraction and utilisation projects in South Africa as registered CDM projects under the Kyoto Protocol. Chapter 6 provides a summary of the main findings of this study.

Sections of this dissertation have been presented at the 11th International Waste Management Symposium (Italy, October, 2009); WasteCon – International Waste Management Conference, (Johannesburg, 2010 and Durban, 2008) and published article in the Clean Air Journal – Bhailall, S., Piketh, S.J., Smith, N., Bogner, J., 2010: Subsurface gas generation at a landfill in Johannesburg, The Clean Air Journal, Vol 18 (1), 10-14. In addition, three papers will be submitted for presentation at WasteCon 2016 to take place in Johannesburg and two papers for publication in Waste Management Research are in progress.

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List of abbreviations

CALMIM	California Landfill Methane Inventory Model
CBPF	Conditional Bivariate Probability Function
CDM	Clean Development Mechanism
CERs	Certified Emission Reductions
CH ₄	Methane
CO ₂	Carbon dioxide
CoJ	City Of Johannesburg
CP1	Commitment Period 1
DEA	Department of Environmental Affairs
DME	Department of Minerals and Energy
DNA	Designated National Authority
DOC	Degradable Organic Carbon
DOE	Designated Operational Entity
DWAF	Department of Water Affairs and Tourism
EMM	Ekurhuleni Metro Municipality
ERU	Emission Reduction Units
EU ETS	European Emission Trading Scheme

FID	Flame Ionisation Detector
FTIR	Fourier Transform Infrared Spectroscopy
IPCC	Intergovernmental Panel on Climate Change
IRD	Initial Rate of Deposition
LFG	Landfill gas
MFMA	Municipal Financial Management Act
MRD	Maximum Rate of Deposition
MSW	Municipal Solid Waste
NEMA	National Environmental management Act
NEMWA	National Environmental Management Waste Act
NO _x	Nitrogen oxide
NWMS	National Waste Management Strategy
OECD	Organisation for Economic Co-operation and Development
PDD	Project Design Document
PPA	Power Purchase Agreement
SAWIS	South African Waste Information System
UNFCCC	United Nations Framework Convention on Climate Change
USEPA	United States Environmental Protection Agency
VERs	Voluntary Emission Reductions

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Chapter 1

Overview

The mechanism of landfill gas (LFG) production and the factors that affect it have been well documented in the literature. This chapter refers to recent research findings on the phases and mechanisms that govern the decomposition of waste and LFG production. Landfill management legislation governing waste disposal in South Africa is discussed. Factors that affect and control the quantity of gas produced and the resultant subsurface gas migration and LFG emissions to the surface are reviewed. Different methods of monitoring subsurface LFG and methods used to estimate LFG production are assessed. Previous studies focussing on landfills in South Africa are also reviewed. In addition, ways of mitigating LFG emissions are considered, with a detailed discussion on the Clean Development Mechanism (CDM) – a flexibility mechanism under the Kyoto Protocol. The current opportunities that exist in South Africa for the reduction of national greenhouse gas emissions through LFG utilisation is also argued. Finally, the main aims and objectives of this study are discussed.

Methane (CH_4) is the most abundant hydrocarbon in the atmosphere and affects the photochemistry of the background troposphere. CH_4 oxidation sets up important reactions that lead to carbon monoxide (CO) and carbon dioxide (CO_2) formation (Bingemer and Crutzen, 1987). The effects of these reactions on tropospheric photochemistry are reliant on concentrations of nitrogen oxide (NO_x) and in environments with high NO_x concentrations. CH_4 oxidation contributes significantly to ozone formation (Crutzen, 1973; Logan et al, 1981; Thompson and Cicerone, 1986; Bingemer and Crutzen, 1987). Increasing concentrations of CH_4 ,

CO and NO_x emissions has resulted in elevated tropospheric O₃ levels since pre-industrial times (Ramaswamy et al., 2001; Dentener et al., 2005).

Sources of CH₄ include biogenic emissions from wetlands, rice production, enteric fermentation from ruminant livestock and waste management (Bingemer and Crutzen, 1987; Gourlay, 1992; Neue *et al.*, 1994; Bogner *et al.*, 1995; Bogner and Matthews, 2003) thermogenic and pyrogenic emissions (fossil fuel and biomass burning). Anthropogenic activities contribute approximately 60% of the total amount of CH₄ emitted each year and landfills contribute less than 5% (1300 MtCO₂e in 2005) of global greenhouse gas emissions (Hein *et al.*, 1997; Houweling *et al.*, 1999, Denman *et al.*, 2007, Bogner *et al.*, 2007).

CH₄ concentrations in the atmosphere have more than doubled during the past several hundred years (Cicerone and Oremland, 1988; Dlugokencky *et al.*, 1998; Shipham *et al.*, 1998) estimated at 722 ppb during preindustrial times in comparison to 1803.2 ppb in 2011 (IPCC, 2001; IPCC, 2007; Hartmann, 2011). This increase has been correlated with an increase in anthropogenic activities in the agricultural sector, energy production, waste management and biomass burning (Denman *et al.*, 2007). Despite CH₄ concentrations decreasing between the early 1980's and 1998 and stabilising between 1999 and 2006, there is a 150% increase in atmospheric CH₄ since 1750. CH₄ concentrations began increasing in 2007 (Hartmann, 2011). CH₄ emissions from agriculture, energy and waste in South Africa was estimated at 51 545 GgCO₂e with emissions from the waste sector contributing to 37.21% of total emissions in the country (DEA, 2014).

Landfills present a unique opportunity to mitigate significant quantities of CH₄. CH₄ emissions from landfills continue for decades after closure of a site (Bogner *et al.*, 2007). Determining the potential success of reduction efforts requires an accurate means of assessing these emissions. Several methods have been established to measure whole landfill CH₄ emissions (Peer *et al.*, 1993; Czepiel *et al.*, 1996b; Tregoures *et al.*, 1999; Galle *et al.*, 2001; Bogner and Matthews, 2003).

A number of LFG estimation models have been developed around the world to estimate emissions from landfills mostly for regulatory and compliance purposes. Most models do not consider site specific characteristics like; material used and thickness of cover soils, climatic conditions and seasonal variability of CH₄ oxidation rates in cover soils (Bogner *et al.*, 2014). However, the models are readily available and are more cost effective than field measurements. Previous studies on landfills in South Africa have focused on the polluting potential of landfills, seasonal variations in waste composition and its effect on LFG production. Other studies have focused on the seasonal patterns in LFG emission rates through field studies using flux chamber measurements. There is a need to accurately estimate annual emissions from landfills in South Africa to quantify its contribution to national greenhouse gas emissions and to aid in the assessment of the possibility of gas extraction systems to mitigate fugitive emissions.

The aims of this research are to quantify and estimate LFG production and measure LFG emissions from landfills in South Africa. LFG estimation models LandGEM, GasSim, the IPCC Waste Model 2006 is used to estimate emissions in all provinces and major Metro Municipalities in South Africa. In order to test LFG estimation

models applicability to landfills in the semi-arid South African environment, LFG emissions are compared to actual gas yields from landfill gas extraction projects at individual landfill sites. These landfills are situated within the City of Johannesburg Municipality and Ekurhuleni Metro Municipality, two of the largest metros in Gauteng. Site specific data, where available, are used in the models to estimate LFG production. In addition, monitoring of subsurface LFG (CH₄, CO₂, O₂, H₂S and CO) concentrations, at five landfill sites, using subsurface gas probes located along the perimeter of the landfill sites has been undertaken to assess LFG quality and migration. This project also measures surface LFG emissions at two landfill sites using flux chambers. Flux emissions are compared to modelled emissions using CALMIM. The air quality impact from a landfill site is measured by the ambient monitoring of LFG using a mobile monitoring station. The contribution of LFG to greenhouse gas (GHG) emissions in South Africa is also assessed and the success of greenhouse gas reductions by applying mitigative strategies and preventing emissions from venting to the atmosphere is evaluated.

Landfill management practices in South Africa

Waste is generated from households, offices, restaurants, public institutions, industry and construction sites, sewage facilities and agricultural activities, and most of the generated waste is landfilled. Landfill management practices are of vital importance with regard to LFG emissions. In many countries, more specifically developing countries, LFG emissions are higher due to improper landfill management practices and inappropriate landfill management legislation. Open dumping and burning is a regular occurrence (IPCC, 2006). Most modern solid

waste management facilities in many countries, including South Africa, consider sanitary landfilling of municipal solid waste (MSW) as the favoured means of waste disposal due to its low cost, compared to alternative methods such as waste treatment and incineration (Morris, 2001). Nevertheless, with South Africa's commitment to sustainable development there is a strong need to meet the broader economic and social challenges of the country whilst protecting the environment at the same time.

Waste management in South Africa faces numerous challenges. According to the Department of Water Affairs and Forestry baseline study report (1998) on the waste profile in South Africa, the mining sector represented the largest fraction of waste generation followed by power generation, agriculture and forestry and industrial waste. Domestic waste only constituted 1.5% of the total waste profile (approximately 13.5 - 15 million tonnes per annum) (DWAF, 1998).

With the promulgation of the National Environmental Management: Waste Act, 2008 (Act 59 of 2008) in 2009, the classification of waste was changed which excludes the mining sector in the current waste profile. Hence, the baseline study conducted by the Department of Environmental Affairs (DEA) in 2012 is not directly comparable to previous baseline studies in 1998 (DEA, 2012).

DEA developed the South African Waste Information System (SAWIS) where detailed information on waste tonnages can be recorded. The accurate collection of waste data is complicated by the lack of information on waste characterisation in South Africa (Sibernagl, 2011; DEA, 2012) and where waste characterisation

information is available it is based on differing methodologies and waste classes, making direct comparisons challenging (Wise *et al.*, 2011; DEA, 2012). Nonetheless, the DEA baseline study of 2012 modelled waste data shows South Africa generated 59 million tonnes of general waste in 2011. An estimated 10% was recycled with the remaining tonnes of general waste going to landfill (DEA, 2012). General waste constitutes 55% (consisting of non-recyclable municipal waste (35%), construction and demolition waste (20%), metals (13%), organic waste (13%) and recyclables (including paper, plastics, glass and tyres (19%)) of the waste profile whilst unclassified waste made up 44% (consisting of brine, sewage sludge, bottom and fly ash, mineral waste and electronic waste) of the waste profile. The remaining 1% is classified as hazardous waste (Figure 1.1) (DEA, 2012).

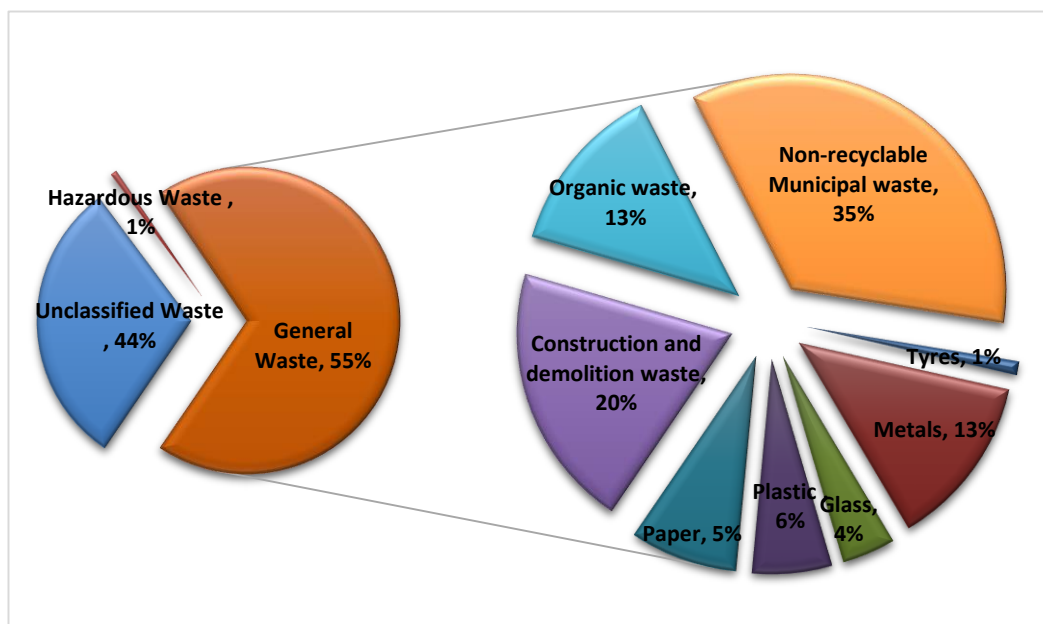


Figure 1.1: Waste profile for South Africa (after DEA, 2012).

Waste Management in South Africa is currently governed by a number of legislative documents, including:

- The South African Constitution (Act 108 of 1996)
- Hazardous Substances Act (Act 5 of 1973)
- Health Act (Act 63 of 1977)
- Environment Conservation Act (Act 73 of 1989)
- Occupational Health and Safety Act (Act 85 of 1993)
- National Water Act (Act 36 of 1998)
- The National Environmental Management Act (Act 107 of 1998)
- Municipal Structures Act (Act 117 of 1998)
- Municipal Systems Act (Act 32 of 2000)
- Mineral and Petroleum Resources Development Act (Act 28 of 2002)
- Air Quality Act (Act 39 of 2004)
- National Environmental Management: Waste Act, 2008 (Act 59 of 2008)
- National Waste Management Strategy, 2011
- National Environmental Management: Waste Amendment Act, 2014 (Act 26 of 2014)

The National Environmental Management Act (NEMA; Act 107 of 1998) provides a legal structure that ensures that the main objectives for environmental policies and decision making in South Africa can be achieved. The principles of NEMA apply to the National Environmental Management: Waste Act, 2008 (Act 59 of 2008) (NEMWA) which came into effect on 1 July 2009, providing framework legislation for the promotion of sustainable waste management practices in South Africa

(Waste Act, 2008). The Act prioritises waste reduction, re-use, recycling, recovery and treatment and as a last resort landfilling (Figure 1.2).

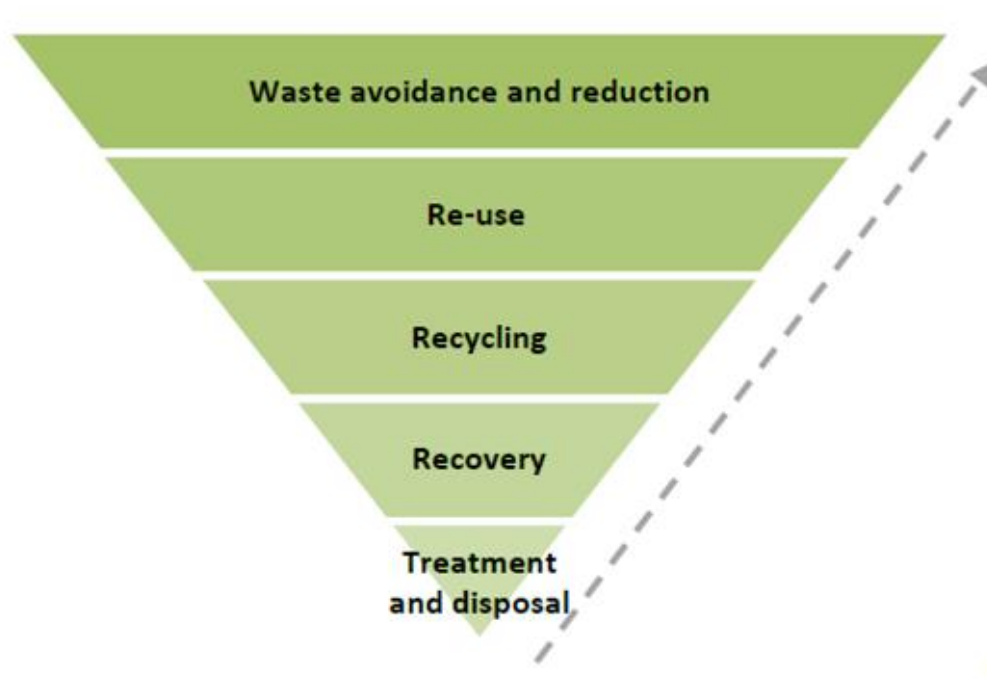


Figure 1.2: Waste hierarchy where waste reduction, re-use, recycling, recovery and treatment is prioritised above landfilling (NWMS, 2011).

NEMWA utilises various legislative tools including the National Waste Management Strategy (2011), the Waste Classification and Management Regulations (2013), Norms and Standards for the assessment of waste for landfill disposal (2013) and Norms and standards for disposal of waste to landfill (2013). The NEMWA was followed by National Environmental Management: Waste Amendment Act 26 of 2014 (“Waste Amendment Act”) which came into effect on 2 June 2014. The National Waste Management Strategy (NWMS) was developed to assist local government in executing the Waste Act and the waste hierarchy in a sustainable way by identifying a number of goals to be executed in order to create a more sustainable waste management future for the country.

According to the NWMS, the main challenges facing waste management in South Africa are:

- An increasing population and growing economy which results in the increase in production of waste which adds pressure on waste management facilities to dispose of the waste generated.
- More complex waste profiles as a result of urbanisation and industrialisation. This mixing of hazardous and general waste adds to the difficulty in managing multifaceted waste streams.
- A backlog of the provision of waste services in urban informal areas as well as rural areas.
- Lack of data as well as data reliability.
- The lack of applicability of the waste hierarchy within waste management.
- The lack of recycling infrastructure that promotes separation at source.
- Archaic waste management infrastructure with limited financial commitment toward operations and maintenance (NWMS, DEA, 2011).

Due to the scarce skills in waste management in South Africa at municipal level the NEMWA and its regulations as well as the NWMS are important guidance documents for municipal officials who manage, maintain and operate disposal sites within a municipality (Mannie and Bowers, 2013). Regulations in these Acts also makes it compulsory to all municipalities to implement and manage an Integrated Waste Management Plan (IWMP) which should be aligned to the Integrated Development Plan (IDP) of the organisation (Mannie and Pretorius, 2013). Before

the Waste Act was issued waste management was governed by a set of documents that set out the minimum requirements for waste disposal. These documents formed part of the permitting system to promote compliance with environmental policy and legislation (Morris, 2001). These documents were used as a guideline for the standard operations for waste disposal in South Africa.

There are three documents that form part of the minimum requirements namely:

1. Minimum Requirements for Handling, Classification and Disposal of Hazardous Waste
2. Minimum Requirements for Waste Disposal by Landfill
3. Minimum requirements for Water Monitoring at Waste Facilities

These documents provide a set of standards and criteria for the selection, investigation, design, preparation, operation, closure and monitoring of landfill sites (DWAF, 1998; Morris, 2001). The methods of waste disposal, gate or weighbridge recording procedures (volumes of waste accepted), hydrological conditions and gas monitoring and management techniques, are key areas to consider when estimating LFG emissions. The location and operation of waste management facilities are dependent on the proximity of the site to human settlements, topography, geology and hydrology (SoER, 2003). According to the Minimum Requirements waste is classified into two broad categories namely general (G) and hazardous (H). For this study only landfills that accept general waste, i.e. waste that does not pose an immediate threat to man and the environment, are considered. General waste consists mainly of MSW which includes a small fraction of garden waste, ash and commercial waste. In addition, some potentially hazardous material such as

batteries, household chemicals and medical waste are often disposed of (DWAF, 1998; Morris, 2001).

Landfills are further classified according to their size and water balance or water capacity and ability to produce leachate. The eventual size of a landfill is dependent on the quantity of waste it will receive over its lifetime which relates to the size of population and area the landfill will service. It is calculated from the maximum rate of deposition (MRD) expected at the site which is dependent on the total amount of waste it receives from the opening year of the landfill (the initial rate of deposition IRD) up to and including its year of closure. It is calculated using the following equation (DWAF, 1998; Morris, 2001):

$$\text{MRD} = \text{IRD} (1 + D)^t \quad (1.1)$$

where MRD (tonne day⁻¹) is the maximum rate of deposition after t years, IRD (tonne day⁻¹) is the initial rate of deposition, D is the expected annual development rate of the landfill based on the expected population growth rate in the area and t is the years since deposition started at IRD.

Table 1.1: Landfill size categories based on the calculated MRD (DWAF, 1998).

Landfill size category	MRD (tonnes per day)
Communal (C)	< 25
Small (S)	25-150
Medium (M)	150-500
Large (L)	> 500

Once the maximum rate of deposition is calculated, landfills are categorised into four sizes namely communal (C), small (S), medium (M) and large (L) (Table 1.1).

The water balance at the site is the third and last level of classification. It determines the potential for significant leachate generation to occur at the site and is affected by rainfall, evaporation, moisture content of waste and water infiltrating the waste body. The climatic water balance can be calculated using a simplified equation (1.2) considering only rainfall and evaporation (DWAF, 1998; Morris, 2001):

$$B = R - E \quad (1.2)$$

Where B (mm) is the climatic water balance, R (mm) is the annual rainfall received at the site and E (mm) is the evaporation from the soil surface. B is calculated from historical rainfall records, working backwards successively (in terms of rainfall volumes) from the wettest year on record. If B is positive for less than one year in five, it is assumed that no significant leachate generation will occur at the site and the site is classified as B⁻ (sporadic leachate generation likely). If, on the other hand, B is positive for more than one year, it is assumed that significant leachate generation will occur at the site and the site is classified as B⁺ (significant leachate generation likely) (DWAF, 1998; Morris 2005). The climatic water balance can be recalculated to take into account the acceptance of waste in liquid form such as slurry or sewage. Most landfills used in this study are classified as GLB⁺ (accept general waste, are of a large size and do produce significant leachate) or GLB⁻ (accept general waste, are of a large size and do not produce significant leachate) sites.

DWAF's Minimum Requirements outlines a basic method of disposal of general waste at large and medium sized landfills. After the deposition of waste in thinly spread layers within the landfill, the waste must be compacted (using a landfill compactor) and at the end of each day the waste must be entirely covered by material (soil) dense enough to isolate the waste from the environment. The cover thickness is dependent on the type of waste and ranges from 0.15 to 0.3 m in thickness (DWAF, 1998).

Different requirements, however, with regard to design and operation of a landfill are dependent on the classification of the landfill. For example, a GL landfill located in Johannesburg will have different design considerations due to its location in a relatively dry climate area (resulting in a B⁻ classification) as opposed to a landfill located in a coastal area of the country with a semi-tropical climate (resulting in a B⁺) classification (Fourie and Morris, 2004).

Whilst this classification is still applicable to landfills in South Africa, a new set of classification was issued according to the NEMWA: Waste Classification and Management Regulations and Norms and Standards for waste disposal to landfill, 2013. These regulations prescribes new landfill classes and liner designs, landfill acceptance criteria and disposal restrictions. It replaces the previous Minimum Requirements for Landfill Disposal (1998) in this regard. There are significant changes to the waste classification and landfill disposal requirements, and sets out new stricter and more stringent barrier designs for Class A, B, C and D landfills instead of G/S/M/LB⁺/B⁻ landfill sites (as shown in Figure 1.3).

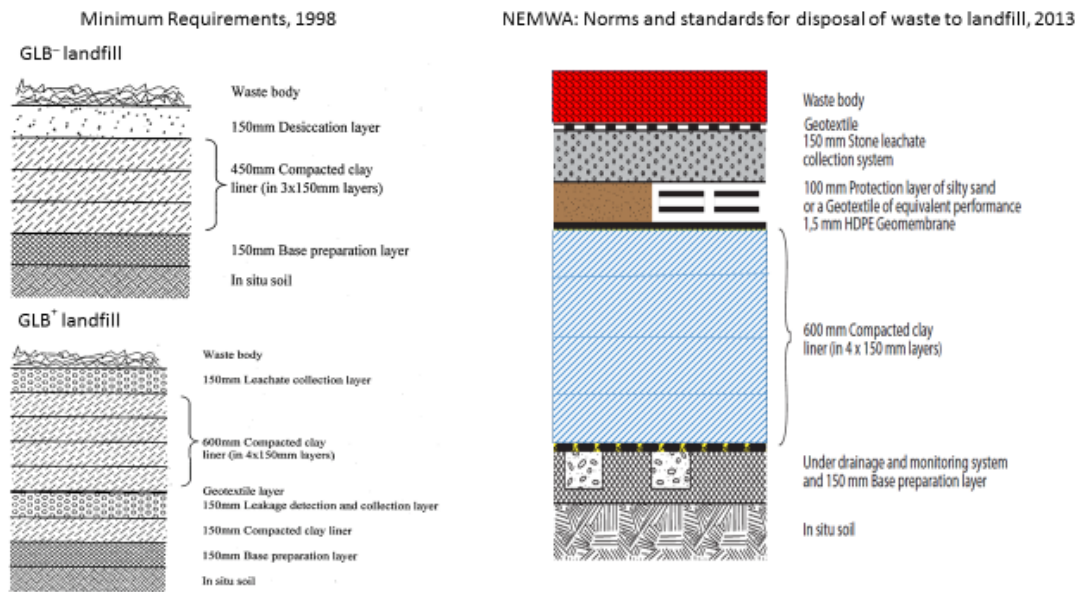


Figure 1.3: Comparison between old landfill classification barrier designs to the new regulations Class B landfill site (DWAF, 1998 and NEMWA, 2013).

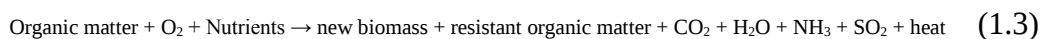
The Minimum Requirements are still applicable for the operation and monitoring of landfill sites. According to the Minimum Requirements LFG must be monitored using subsurface probes and CH₄ analysers, at intervals of three months and emissions managed at all landfill sites. The standard for gas concentrations is 1% by volume (v/v) at standard temperature and pressure (STP). Gas monitoring should continue after closure of the site until the gas concentrations are at a safe level (DWAF, 1998). Presently only a few gas collection and utilisation systems exist in South Africa. Passive venting to the atmosphere are the most common practices by both municipalities and privately owned landfill sites (Bogner and Lee, 2004). LFG emissions from landfills are becoming subject to strict regulatory controls in many developing countries (Bogner and Spokas, 1993). This is done by monitoring gas emissions using subsurface probes and by modelling emissions.

Landfill processes and gas production

Waste degradation and landfill gas production processes

The physical (specific weight, moisture content, particle size distribution, field capacity, porosity and compactibility), chemical (energy potential and nutrient availability) and biological (available solids such as sugars, fats and proteins) properties of waste are all transformed in some way during waste degradation (Tchnobanoglous et al, 1993; Morris, 2001). Changes in the biological processes (aerobic or anaerobic) of waste are more significant with regard to LFG production than the physical or chemical processes (Morris, 2001).

At the time of waste deposition in a landfill, oxygen is present between waste materials resulting in the occurrence of aerobic degradation processes where aerobic bacteria rapidly convert biodegradable organic matter to produce CO₂, water, and other by-products (Tchnobanoglous et al, 1993; El-Fadel et al, 1997; Moriss, 2001).

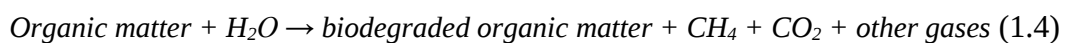


This process is affected by time, the composition of the waste, moisture content, availability of nutrients and other environmental factors (Thorneloe *et al.*, 1993; Morris, 2001). Once all oxygen present is depleted by aerobic bacteria, anoxic conditions are established within the landfill. The next stage of decomposition is then reached and is referred to as the anaerobic decomposition stage. The commencement of anaerobic decomposition requires several months to years after

waste disposal (Thorneloe *et al.*, 1993). Figure 1.4 represents the steps of LFG production during anaerobic decomposition of waste. These steps are highly inter-dependent and the complex reactions depend on several groups of bacteria including fermentative, acetogenic and methanogenic bacteria (Christensen *et al.*, 1996, El-Fadel *et al.*, 1997; Morris, 2001).

In the first step (hydrolysis), hydrolytic reactions break down organic matter from waste to simpler forms such as amino acids, simple sugars, fatty acids and other low molecular weight organic compounds. Acidogens convert the products of hydrolysis (by acid fermentation) to organic acids and alcohols. These products can then be utilised by acetogenic bacteria in the next step (acetogenesis) converting them to acetate, hydrogen (H₂) and CO₂ (Christensen *et al.*, 1996; Czepiel *et al.*, 1996). In the final step methanogenic bacteria utilises the acetate and H₂ as energy sources to produce CH₄.

A generalized chemical reaction for the anaerobic decomposition stage is given as (Tchobanoglous *et al.*, 1993; Christensen and Kjeldsen, 1996; Morris, 2001):



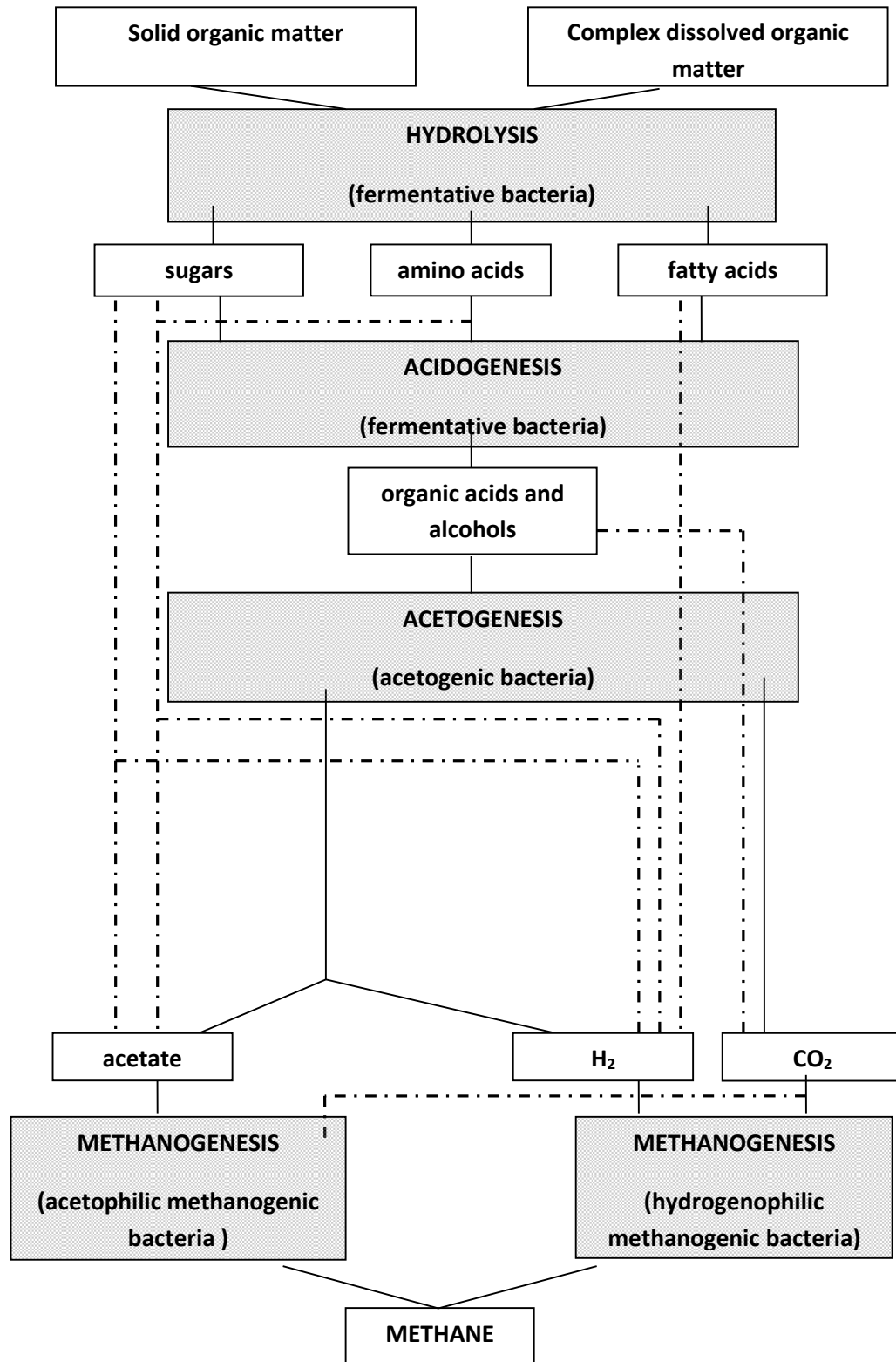
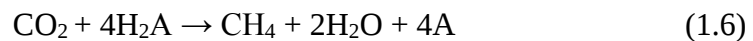


Figure 1.4: Anaerobic degradation sequence of waste leading to the formation of CH₄ (after Morris, 2001 and Scharff et al, 2005). Dotted lines represent intermediate reactions and solid lines main reactions leading to methanogenesis.

The final step of methanogenesis can be broken down further and is characterised by two types of methanogenic bacteria. Acetophilic bacteria convert acetic acid (by decarboxylation) to CH₄ and CO₂ shown in reaction (1.5).

This reaction produces the largest fraction of CH₄ in LFG and also results in the formation of a large amount of CO₂. Reaction (1.6) shows the conversion of CO₂ and H₂ to CH₄ by hydrogenophilic bacteria with organic molecules (H₂A) as hydrogen donors (Christensen et al, 1996; Czepiel *et al*, 1996).



The overall influence of these reactions has been shown to occur sequentially in phases (Figure 1.5). The development of these phases is based on the assumption that, upon placement of the waste, aeration does not occur and that conditions within the landfill are sufficient to encourage and to maintain CH₄ production (Farquhar and Rovers, 1973). These phases are described below according to Farquhar and Rovers (1973), Christensen and Kjeldsen (1996) and Morris (2001). However, it is important to note that these phases are not homogenous and due to local variations in waste composition, reaction rates can differ significantly from place to place and all phases can occur simultaneously throughout the landfill.

Phase 1 represents the beginning of degradation that is, the aerobic stage when available oxygen in the landfill is depleted by aerobic bacteria to produce mostly CO₂ and water. The soil used as a daily cover is the main source of both the aerobic

and the anaerobic organisms responsible for waste degradation. This phase is fairly short and only occurs for a few days.

Phase 2 occurs with the onset of anaerobic conditions. At first, nitrate and sulphate are reduced to nitrogen and hydrogen sulphide followed by production of H_2 , CO_2 and methane (with a reduction in the redox potential) through the four-step process of hydrolysis, acidogenesis, acetogenesis and methanogenesis. Due to the high concentrations of CO_2 and the production of fatty acids there is also a drop in pH.

Phase 3 is characterised by the production of large amounts of organic acids and smaller quantities of H_2 mainly due to the acceleration of microbial decomposition processes initiated in phase 2.

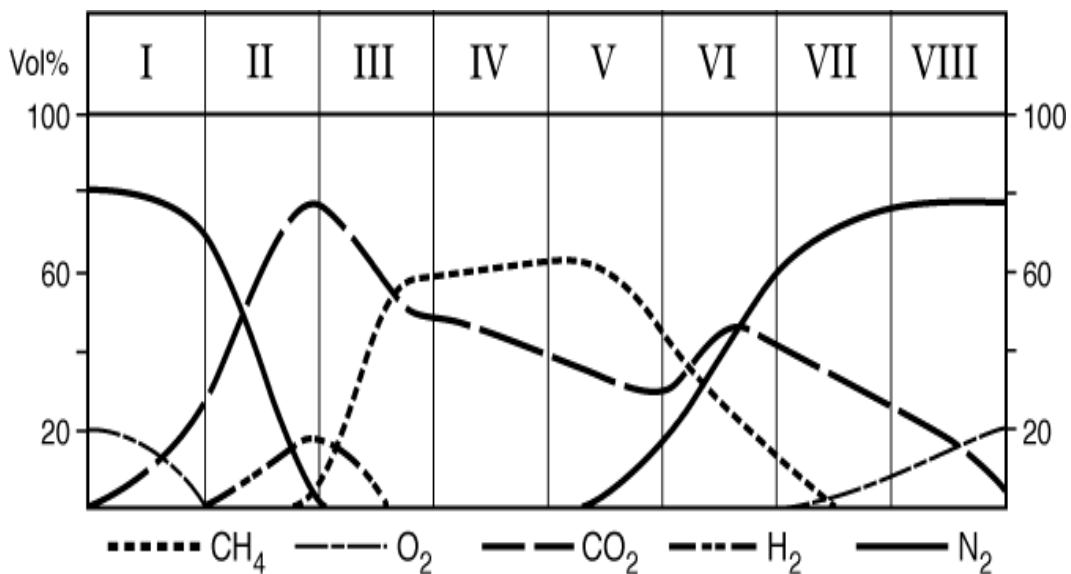


Figure 1.5: Phases of landfill gas production with waste decomposition (Farquhar and Rovers, 1973).

Phase 4, also referred to as the maturation phase or methanogenic phase, is characterised by 50 – 60 % CH_4 and low concentrations of H_2 which is oxidised by

CO₂ to CH₄. During **Phase 5** CH₄ production decreases as air starts to intrude into the outer parts of the landfill body. During **Phase 6** there is an increase in CO₂ as CH₄ migrating from the centre of the landfill to the top of the landfill body is oxidised.

During **Phase 7** CH₄ formation is insignificant due to intruding air oxidising solid organic carbon to produce more CO₂. At the final phase, **Phase 8**, the rates of gas production processes resemble the rates found in an active soil layer.

After the stabilisation of anaerobic conditions in Phase 2 within a landfill site, the LFG formed consists of approximately CH₄ (45 – 60 %), CO₂ (40 – 60 %), nitrogen (N₂) (2 – 5%), O₂ (0.1 – 1 %), H₂ (0 – 0.2 %) and CO (0 – 0.2 %). The remainder of the gas is composed of over 200 trace components, generally non-methane hydrocarbons (Bogner *et al.*, 1990; Augenstein and Pacey, 1991; Tchnobanoglous *et al.*, 1993; Morris, 2001; Stratchan, 2004, IPCC, 2006) (Table 1.2).

There are no time estimates associated with the above mentioned sequence owing to their dependence on a number of factors including the amount of organic compounds in a landfill, nutrient availability and moisture content and availability. After the initial aerobic phase which lasts for a few days or weeks the other phases may occur over months, years and even decades. The phases can even cease for long periods when conditions are not suitable for the processes to proceed (Christensen and Kjeldsen, 1996; and Morris, 2001).

Table 1.2: Characteristic landfill gas constituents and their concentrations (Tchobanoglous et al, 1993 and Pitchel, 2005) for a generic landfill site.

Component	Percent (dry volume basis)
Methane (CH ₄)	45-60
Carbon dioxide (CO ₂)	40-60
Nitrogen (N ₂)	2-5
Oxygen (O ₂)	0.1-1.0
Sulphides, disulphides, mercaptans, etc	0-1.0
Ammonia (NH ₄)	0.1-1.0
Hydrogen (H ₂)	0-0.2
Carbon monoxide (CO)	0-0.2
Trace constituents	0.01-0.6
Characteristics	Value
Temperature, °C	38-50
Specific gravity	1.02-120
Moisture content	Saturated
High heating value, kJ/m ³	900-1100

Table 1.3 shows the change in LFG composition 48 months after the closure of a landfill. The volume of these gases released can be estimated in a number of ways. Degradable organic carbon (DOC) is an important parameter affecting CH₄ emissions. It is based on the waste composition and varies for different waste fractions.

Accurate estimates of the amount of waste and amount of DOC in waste can be done by sampling the waste accepted at the landfill and measuring DOC before deposition or specifying the waste stream for each waste type and source. Food, garden waste, paper and cardboard contain most of the DOC in waste (IPCC, 2006).

Table 1.3: Methane and carbon dioxide concentrations observed during the first 48 months after the closure of a landfill cell (Tchobanoglous et al, 1993 and Pitchel, 2005) for a generic landfill site.

Time interval since cell completion (months)	Average (percent by volume)	
	Methane (CH ₄)	Carbon dioxide (CO ₂)
0 – 3	5	88
3 – 6	21	76
6 – 12	29	65
12 – 18	40	52
18 – 24	47	53
24 – 30	48	52
30 – 36	51	46
36 – 42	47	50
42 – 48	48	51

Assuming that the biodegradable organic waste is completely converted to CH₄ and CO₂, the total volume of gas can be estimated using the following equation (Buswell and Mueller, 1952; Tchobanoglous et al, 1993; Christensen and Kjeldsen, 1996; Morris, 2001; Pitchel, 2005):

$$C_aH_bO_cN_d \left(\frac{4a - b - 2c + 3d}{4} \right) H_2O \rightarrow \left(\frac{4a + b - 2c - 3d}{8} \right) CH_4 + \left(\frac{4a - b + 2c + 3d}{8} \right) CO_2 + dNH_3 \quad (1.7)$$

By varying the assumptions on which the above formula is based, a number of gas yields from MSW degradation can be estimated (Morris, 2001). Using an organic carbon component of 50% of waste (in terms of dry mass) and 90% biodegradability of this carbon, lysimeter studies by Shamrock (1998) estimate a theoretical yield of

460 m³ CH₄ (dry tonne⁻¹) and a gas composition of 55% CH₄ and 45% CO₂. Field measurements in the same study found actual yields of 30 – 180 m³ (dry tonne⁻¹). The study concluded that conditions in the landfill were not optimal for bacterial growth and that a large amount of the organic carbon in the MSW is not available for degradation as it is washed out in leachate (Morris, 2001).

Tchobanoglous et al (1993) presented another method of calculating LFG yields in which the organic material present in MSW is divided into rapidly decomposing (three months to five years) and slowly decomposing materials (up to 50 years and more). The percentage distribution of the major elements C, H, O and N is then computed for both material groups. A chemical formula is used (by normalising mole ratios a, b, c and d) in equation 1.7 using N as a reference with a mole ratio of 1 (Morris, 2001).

As mentioned there are a number of environmental or abiotic factors and different landfill operation techniques that influence the rate of waste decomposition in a landfill. The relationships between these factors are discussed in the following section. More detailed descriptions can be found in Christensen and Kjeldsen (1996) and Morris (2001).

Controls on waste degradation and landfill gas generation

LFG production and emission rates depend on the inter relationship of many complex variables (Figure 1.6). The major variables which are expected to influence the volume and rate of emissions are waste composition, temperature,

moisture content, method of disposal, number of microbial populations present within the landfill, available nutrients, time and pH (Beck, 1993).

The total amount of waste in place and its composition are important considerations when estimating the CH₄ generation potential from a landfill. Waste composition and compaction play a significant role in the rate at which MSW decomposes. The composition of waste is usually dependent on the types of waste used by the community the landfill serves. The most important aspect is the carbon content or DOC present in the waste. Carbon is one of the three major elements needed for optimum microbial activity and development. Therefore, the percentage of waste consisting of biodegradable organic compounds is important with regard to the rate of degradation in a landfill (Tchnobanoglous *et al.*, 1993; Pitchel, 2005).

Shamrock (1998) investigated the effect of waste composition on the polluting potential of a landfill. A detailed compositional analysis on landfills in South Africa from different communities was completed in the study. By means of laboratory lysimeters (over a period of 264 days) and a number of small and large scale field cells it was concluded that changes in waste composition do affect LFG polluting potential. Lysimeters are used to simulate a “model” landfill as a means to simulate actual landfill conditions. A similar study by Morris (2001) found that stable methanogenic conditions can be achieved faster in landfills containing waste with at least some ash content than without. It showed that some waste mixing can enhance the degradation processes within a landfill.

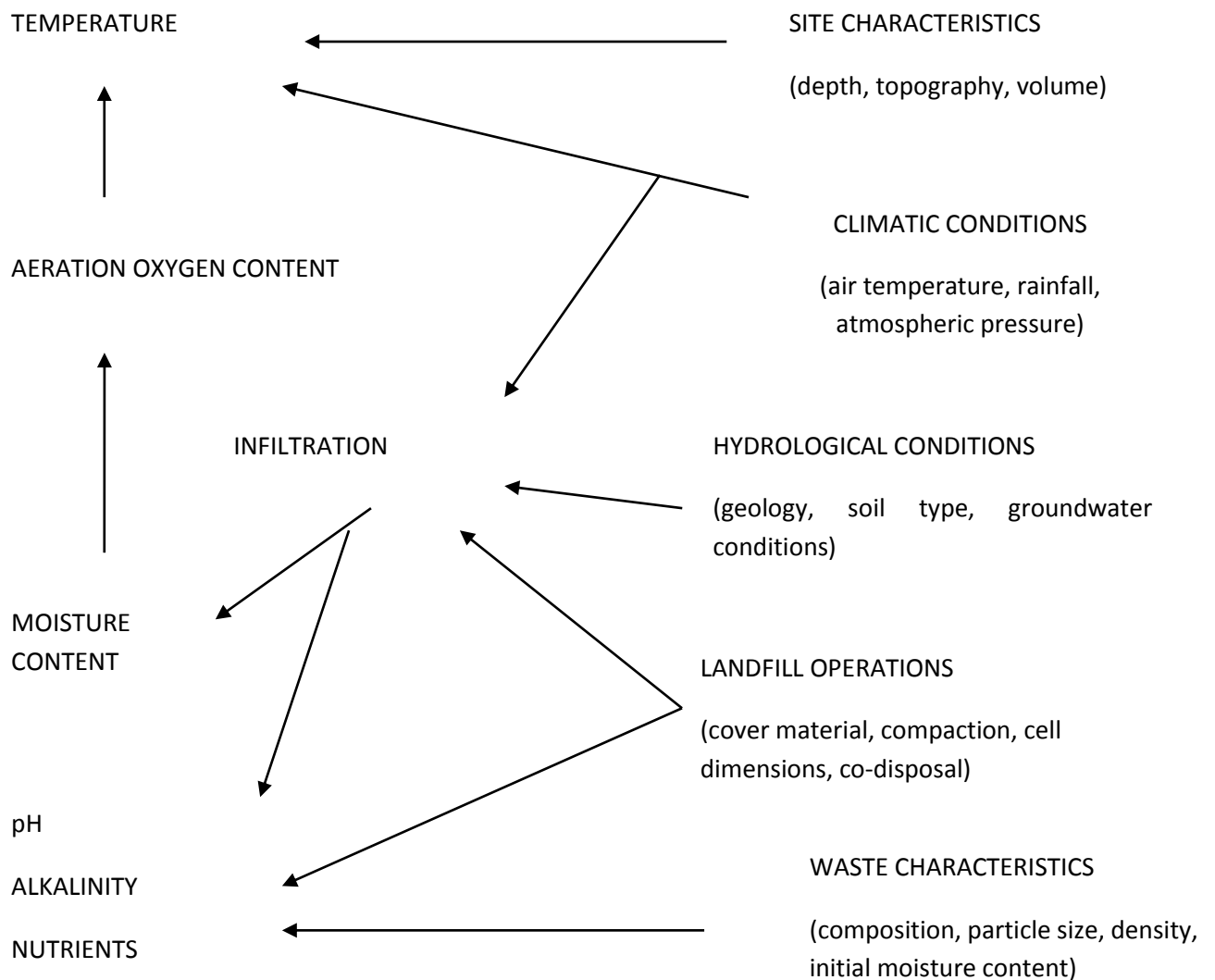


Figure 1.6: Relationship between factors affecting rates of MSW decomposition in landfills (Morris, 2001).

Co-disposal or the addition of sewage and sludge to a landfill also aids in waste degradation mainly due to increasing water content. Moisture within a landfill is essential since all phases of waste degradation take place in an aqueous state. Several laboratory studies have shown that LFG formation increases with increasing moisture content within the landfill. With higher water availability microbial metabolism is accelerated until saturation. As a result, CH_4 and total gas production is increased within the landfill (Bogner, 1990; Bogner, 1993).

It is important to note however, that if the methanogenic phase in the landfill is already established, the addition of sewage sludge may have a minimal effect on decomposition. The pH of the sewage sludge also affects the rate of degradation. pH has a significant influence on LFG production since methanogenic bacteria are only active within a limited pH range of 6.5 – 8 although the pH range for fermentative and acetogenic bacteria is much wider. If the pH of the added sewage sludge is low then there is a decrease in LFG production whilst neutral sewage sludge can increase gas production (Tchnobanoglous *et al.*, 1993; Pitchel, 2005; Morris, 2011).

Compaction of waste on a daily basis aids in LFG formation especially with dry waste as it increases the moisture content and reduces the intrusion of air within the landfill. In addition, a daily soil cover can also increase LFG production if the soil used provides important buffer capacity to the landfill allowing for optimum pH values for CH₄ production. Nevertheless, the capping of waste can also lower LFG formation. When the soil layer undergoes aerobic decomposition it decreases the diffusion of oxygen to the landfill thus decreasing the composting rate. However, the absence of oxygen and low redox potentials are essential for bacterial activity and LFG production (Tchnobanoglous *et al.*, 1993; Pitchel, 2005).

The infiltration and recirculation of leachate has both a negative and a positive effect on LFG production. The rate of moisture infiltration through a landfill cover affects the moisture availability to the landfill. Leachate recirculation can be used in dry climates as it increases the water content in the waste, supplies additional nutrients and aids in diluting high concentrations of inhibitors. The presence of

inhibitors such as heavy metals has toxic effects on methanogenic bacteria and inhibits the formation of LFG (Tchnobanoglous *et al.*, 1993; Pitchel, 2005).

LFG formation is highly affected by temperature. At elevated temperature gas production is more vigorous and produces more heat and as such is self-enhancing. Elevated temperatures primarily develop in landfills with a good CH₄ production, a moderate water flux and substantial waste thickness providing good insulation. Lysimeter studies show that methanogenic activity increases significantly (up to two orders of magnitude) with temperature increases from 20°C to 40°C with optimal temperature reported in the range of 34°C to 51°C.

The presence of H₂ and sulphates is also important for LFG formation. H₂ is formed by processes in the landfill and is also used as a substrate. During fermentation a low partial pressure of H₂ leads to the formation of acetate and CO₂ whilst a high partial pressure does not form these products, leading to high concentrations of fatty acids and a subsequent decrease in pH. When sulphate concentrations are high the production of LFG is reduced due to substrate competition by sulphate-reducing bacteria and methanogenic bacteria (Tchnobanoglous *et al.*, 1993; Pitchel, 2005).

Many modelled estimates of LFG emissions are likely to under-estimate actual emission rates because of aerobic oxidation of CH₄ near the surface by methanotrophs (El-Fadel *et al.*, 1997). The breakdown of CH₄ by methanotrophic bacteria to form CO₂ occurs as follows:



Field studies have shown that a significant portion (30 – 100%) of the CH₄ present in cover soils is oxidised to CO₂ by methanotrophic bacteria within the soil cover (Bogner *et al.*, 1995; Mancinelli, 1995; Czepiel *et al.*, 1996a; Whalen *et al.*, 1990; Christopherson *et al.*, 2000). This process is also affected by climatic conditions, for example, experiments with ¹³C have shown that no CH₄ oxidation occurs at atmospheric temperatures below 0°C (Chanton *et al.*, 1999, Börjesson *et al.*, 2000). In addition, these studies also show that if the CH₄ flux is less than the CH₄ oxidizing capacity of the landfill cover, the soil cover bacteria oxidises atmospheric CH₄ as it diffuses into the soil (Bogner *et al.*, 1995).

Variations in atmospheric pressure due to large weather systems, for example cold fronts and inter tropical convergence zone formations, intensify landfill gas migration (Lombard, 2004). Czepiel *et al.*, (2003) show that a barometric pressure decrease of 10 hPa causes a tripling of the CH₄ emissions from a landfill due to a change in pressure gradient between the landfill and the atmosphere. However, this effect will be site specific and is an important factor that needs to be taken into consideration with landfills in South Africa.

Surface LFG emissions are highly variable both spatially and temporally, varying on a daily basis due to changes in air pressure and rainfall which affect the permeability of the top layer. Seasonal variations in surface emissions occur due to reduced oxidation in cover soils in winter. In addition, due to differences in waste amounts, age and composition in different parts of the landfill, emissions vary over the surface. The slopes of a landfill generally have higher emissions than the upper surface due to high horizontal permeability compared to vertical permeability.

Emissions also vary due to regions of reduced permeability in the subsurface and due to cracks in the surface. As a result, emissions over a few metres can vary over a factor of 1000.

In many countries, more specifically developing countries, LFG emissions are higher due to improper landfill management practices, open dumping and inappropriate legislation governing the management of landfilled waste. Landfill management practices are of vital importance with regard to gaseous emission rates.

Environmental impacts associated with landfills

Landfills are the most prevalent method of waste disposal worldwide. This is mainly due to the low cost associated with landfilling as compared to other disposal systems. An ever increasing population and high rates of urbanisation will ensure that landfilling will continue to be the most common method of disposal (Bogner *et al.*, 1995).

The migration of LFG and leachate away from landfill boundaries and their release into the surrounding environment can result in serious pollution concerns, the most important of which are the effect on air and water quality (El-Fadel *et al.*, 1997). It can also lead to various other hazards such as fire and explosion hazards, health risks and can even cause damage to vegetation (Kjeldsen, 1996).

The solubility of LFG in water leads to leachate production that can contribute to surface and groundwater pollution. Emissions due to waste decomposition contribute to air pollution and also create nuisance problems due to odours

(Kjeldsen, 1996; Morris, 2001; SoER, 2003). The unpleasant odours associated with landfills originate from the initial fermentation stage of the waste within the landfill. Odours are not linked to the primary constituents of LFG i.e. CH₄ and CO₂, as both these gases are odourless (Kjeldsen, 1996).

LFG is explosive mostly due to its CH₄ content. CH₄ is explosive at concentrations of 5 – 15 % in air. Venting of the gas to the atmosphere prevents any explosion risk; however, the concern lies with the lateral migration of CH₄ through soil and along cracks and its subsequent accumulation (Kjeldsen, 1996; Morris, 2001). This highlights the importance of subsurface LFG monitoring.

The most significant environmental problem associated with LFG emissions pertaining to this study is its contribution to the increase of global greenhouse gases in the atmosphere. Due to their large numbers, landfills have become a large contributor to atmospheric CH₄ concentrations (El-Fadel *et al.*, 1997). CH₄ contributes to global warming as a result of its ability to trap heat 21 times more effectively than CO₂ over a 100 year period (IPCC, 2001). This was changed to 25 times by the Working Group I to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change. Hence LFG extraction project will use a GWP 25 times greater for CH₄ emissions from 2013 onward (IPCC, 2007; COP/MOP 18, 2012)

The total positive climate forcing attributed to CH₄ over the last 150 years is 40% of that of CO₂ (Hansen *et al.*, 1998). Decomposition of organic material derived from biomass sources (crops, wood) is the primary source of CO₂ emissions from

landfills and is considered as greenhouse effect neutral as the carbon is of biogenic origin (Huber-Humer *et al*, 2004 and IPCC, 2006).

The global anthropogenic emission of CH₄ is about 360 TgCH₄/a (Wuebbles and Hayhoe, 2002) and landfill CH₄ emissions are the largest contributor (Bogner and Spokas, 1993). In many developed countries landfills are estimated to account for approximately 10 – 19% of annual anthropogenic greenhouse gas emissions (Stern and Kaufmann, 1996; IPCC, 2001; Huber-Humer *et al*, 2004). Many surveys in the literature predict that the annual volume of CH₄ generated in many developing countries from landfills will increase significantly over the next decade. This is mainly due to the expansion of forced landfilling practices to deal with large quantities of waste especially in rapidly growing urban centres (Spokas *et al.*, 2006).

Landfill CH₄ contributes and will continue to contribute significantly to global greenhouse gas emissions. Therefore, estimating and quantifying CH₄ emissions from landfills is essential if we are to evaluate measures for reductions in national and global greenhouse gas emissions. According to the IPCC, a reduction in total global CH₄ emissions of approximately 10% would stabilise concentrations at current levels (IPCC, 2001).

Landfills offer a concentrated source of LFG and more specifically CH₄ emissions from landfills can be controlled or mitigated. Since CH₄ has an atmospheric lifetime of about ten years reductions in individual CH₄ sources can decrease atmospheric concentrations within a decade (Houghton *et al.*, 2001). CH₄ emission reduction

should therefore be part of a climate policy portfolio, and this is indeed recognised in international negotiations, most notably the Kyoto Protocol.

CH₄ emissions from the waste sector contributed to 37.21% of total CH₄ emissions in South Africa in 2010. South African waste management act and regulations do not require that landfill gas be recovered or utilised hence any LFG that is captured and flared or utilised preventing it's venting to the atmosphere will constitute a reduction in greenhouse gas emissions (Bogner and Lee, 2004).

There is a need to estimate annual emissions from landfills in South Africa to aid in the assessment of the possibility of gas extraction systems for landfills. This will allow the country to exploit economic resources available under the Kyoto Protocol through the Clean Development Mechanism (CDM) and possibly reduce national CH₄ emissions.

In order to assess the risks and hazards associated with LFG it is important to know the variation in LFG generation both seasonally and annually and how changes in meteorological conditions affect the production of LFG. It is also of vital importance that LFG emissions be estimated at individual landfills to aid in assessing the possibility of gas extraction and utilisation.

Subsurface gas migration

LFG does not only diffuse to the surface of the landfill and disperse into the atmosphere but it also migrates within the landfill and is governed by a number of factors. There are a number of conditions that forces LFG to migrate laterally within

the landfill body rather than vertically. Most importantly there must be a driving force that moves the gas and an available pathway (Hooker and Bannon, 1993; Pitchel, 2005).

CH₄ and CO₂ migrate as a function of the density of the gas mixture and other parameters such as temperature and partial pressure. LFG will migrate through a path of least resistance (Tchobanoglous, 1993; Pitchel, 2005). The gas migration is also controlled by many landfill operation processes including the placement of the waste and the permeability of the daily cover material (Pitchel, 2005).

At most landfills waste is commonly placed in a layered formation which aids in the production of horizontal channels within the landfill. The lateral migration of gases is further aided by the compaction of the waste which results in lower porosity primarily at the surface of each waste layer (Kjeldsen, 1996). If the daily cover soil is dense or impermeable, such as a clay soil, lateral migration is favoured. Additionally, with an increase in soil moisture content in the cover, vertical LFG flow is further inhibited (Pitchel, 2005).

Several meteorological factors also affect the migration of LFG, the most important of which are changes in the barometric pressure over time. At most active landfills, there is often a high variability in the gas pressure within the landfill. However, the internal pressure is usually greater than atmospheric pressure. This results in convective (pressure-driven) and diffusive LFG flow (Tchobanoglous et al, 1993; Kjeldsen, 1996).

All these factors together control the maximum distances that LFG can migrate laterally. For example, coarse (porous) soils such as sand and gravel will promote lateral transport of gases, at greater distances away from the landfill boundary, than would fine grained soils (Pitchel, 2005). Very few studies have investigated the maximum distances to which the LFG can migrate; however, it is believed that the migration has a high spatial and temporal variability (Kjelsen, 1996).

In order to ensure that LFG does not migrate to structures such as buildings on the landfill site and its surroundings, it must be regularly monitored. The frequency of LFG monitoring is determined by identifying the possible gas pathways based on soil conditions, geology, hydrogeology and location of facility structures (Hooker and Bannon, 1993; Pitchel, 2005). The frequency of monitoring should be sufficient to detect LFG migration based on changing landfill conditions. Gas sampling probes placed within the landfill body is the most common method used to monitor subsurface LFG. Probes can either be temporary (search bar holes and disposable or removable probes) or permanent (trial pits, boreholes, fixed-in-place probes and gas wells) (Hooker and Bannon, 1993; Pitchel, 2005).

They can be single level, multilevel or installed in clusters with samples collected by active or diffusive processes. The number, type and location of gas probes is site specific and dependent on subsurface conditions, land use and location and design of facility structures (Pitchel, 2005). Details on the installation of permanent structures are given in Farqurah (1997). Methods for measuring pressures, flows and gas concentrations in boreholes and other facilities are outlined in Hartless (1995) and Straka *et al.*, (1997).

Quantifying landfill gas emissions

As discussed landfills present a unique opportunity to mitigate significant quantities of CH₄ because they are large, concentrated sources and the gas recovery technology is fairly simple and has economic value. However, to determine the potential, effectiveness and success of efforts aimed at reducing CH₄ emissions from landfills it is important to assess and quantify these emissions (Scharff and Jacobs, 2006). LFG emissions are important from both a regulatory, due the significant contribution of CH₄ to global warming, and a financial perspective, due to a growing market for CH₄ emissions credits. This highlights the need for accurate methods to calculate the generation and emissions potential of individual landfills (Czepiel *et al.*, 2003). The estimated amount of CH₄ emitted due to anaerobic decomposition of waste can be based on theoretical models, laboratory studies and field measurements (Peer and Thornloe, 1993).

Direct field measurements of landfill gas emissions

Gas emissions from landfills have a high temporal and spatial variability (Scharff *et al.*, 2000; IPCC, 2006). Therefore, it is not easy to measure CH₄ emissions. Nevertheless, a number of methods are described in the literature (Table 4). Emissions can be calculated from concentration profiles in soil cores, for example as in Bogner and Scott (1995); spatially distributed static and dynamic closed chambers can be used (Bogner and Scott, 1995; Cossu *et al.*, 1997; Maurice and Lagerkvist, 1997; Cioni *et al.*, 2003; Capaccioni *et al.*, 2005; Raco *et al.*, 2010; Asadi *et al.*, 2013; Giovenali *et al.*, 2013) mass-balance or micrometeorological

methods can give concentration profiles on top of the landfill from which whole landfill emissions can be determined (Oonk and Boom, 2000, Savanne *et al.*, 1997); and measurements at a distance from the landfill (using a tracer gas to detect gas plumes) can be used to obtain emissions from the entire landfill (Czepiel *et al.*, 1996b; Scharff and Hensen, 1999). Methods of assessing landfill gas emissions are site specific and require a proper evaluation of the site before monitoring.

Previous studies by Morris (2001) and Fourie and Morris (2004) on landfills in South Africa used a static chamber technique. Chambers can be operated statically or dynamically. Static closed chambers consist of a box with a surface area approximately 0.5 m² in which the CH₄ fluxes are measured. Dynamic closed chambers are similar to static chambers except that continuous air flow is maintained through the chamber (using inlet and outlet fans) (Scharff *et al.*, 2001). Thus, due to the lack of power supply at the landfill sites, dynamic chambers were considered difficult to implement and static chambers were chosen for this study. Measuring whole surface emission rates can be problematic when using limited discontinuous surface chambers (Spokas *et al.*, 2003) due to the heterogeneity of flux measurements across the surface of the landfill (Bogner *et al.*, 1995; Cardellini *et al.*, 2003; Jones and Nedwell, 1993; Mosher *et al.*, 1999). In addition, the method is very labour intensive as it can take several days to collect enough data to obtain whole landfill emissions. This method is also unable to measure CO₂ emissions on surfaces with vegetation and can only be used for shorter periods (Scharff *et al.*, 2001). The advantages are that it is low cost and requires a low level of technology for its construction, operation and maintenance, and it also has a rapid data turn-around. Most current worldwide understanding of LFG emissions has resulted from

the use of chambers (Morris, 2001). A number of ways have been identified to improve the accuracy or reduce the number of measurements, for example using geostatistical methods to expand the measurements from a small section on the landfill to represent the whole landfill (Börjesson *et al.*, 2000; Spokas *et al.*, 2003) or identifying the main emitting zones (cracks, areas where vegetation is under stressed and edges between capped areas and slopes) (IPCC, 2006).

Micrometeorological methods measure emissions as a flux through an imaginary horizontal plane and this is recalculated as vertical fluxes (IPCC, 2006). Gas fluxes are determined by assuming that gas flux densities measured in the atmosphere represent the gas flux from the cover soil. This assumption is valid if the soil cover is uniform (extending for a distance of 75 – 100 times the flux measurement height) and the fluxes are steady with time. These methods are automated and can provide diurnal and seasonal patterns in emissions (Morris, 2001).

These methods are better suited to measure emissions from relatively large landfill surfaces during longer times (seasonally or annually) with minimal disturbance to the underlying surface. A disadvantage of this method is that the instrumentation used is highly sophisticated and expensive and results are obtained through very complicated calculations (Bogner *et al.*, 1997b; Maurice, 1997). Some commonly used methods include the gradient and eddy-covariance methods (Tregoures *et al.*, 1999; Bogner and Scott, 1995; Laurila *et al.*, 2005) and the mass balance method (Maurice, 1997; Tregoures *et al.*, 1999; Fowler and Duyzer, 1989; Oonk and Boom, 2000). The method can provide emission estimates of CH₄ and CO₂ with high spatial and temporal resolution. In addition, the method's capability to measure both

CH₄ and CO₂ provides a data set that gives insight to the primary processes leading to emissions such as CH₄ formation and oxidation (Scharff *et al.*, 2001). Atmospheric influences including rainfall, wind and ambient air pressure all result in variations of emissions in time and are important measurements that need to be considered when using this technique (Czepiel *et al.*, 1996a).

Tracer or plume measurement techniques rely on simultaneous CH₄ measurements and an inert gas, usually SF₆, (released at a known rate) from a source or a number of sources from the landfill. The magnitude of the gases fluxes is related to the ratio between the measured concentrations of the two gases. Concentration measurements are made at several points downwind of the plume (Czepiel *et al.*, 1996a; Bogner and Scott, 1995; Galle *et al.*, 2001; Scheutz *et al.*, 2013). Some disadvantages of this technique are the need for high CH₄ concentrations (above background levels) and the fact that CH₄ from various other sources can interfere with measurements. The method depends on meteorological conditions, and sampling needs to be conducted at great distances (greater than 100 m depending on the size of the landfill) from the landfill to ensure adequate mixing of the tracer gas in the gas plume (Morris, 2001). Other plume measurements include the use of tunable diode laser (TDL) or FTIR technology. However, the method is very expensive and is normally only applied for one or a few specific days which makes it unsuitable to monitor annual emissions. CH₄ isotope measurements are widely recognised for their applicability in quantifying the amount of methane oxidised in cover soils and can be used to validate the accuracy of the mass-balance or stationary plume method by calculating the effect of methane oxidation on emissions (NOVEM, 2003).

Table 1.4: Comparison of different methods used to measure LFG emissions (NOVEM, 2003).

Method	Spatial resolution	Temporal resolution	Component	Costs	Experiences (world wide)	Other advantages/ draw-back limitation
Soil core	m ²	hour	CH ₄ and CO ₂	high	Few	Especially suited for mechanistic studies of oxidation, possible interference with normal landfilling activities
Closed chambers	m ²	hour	CH ₄	low	Many	Many samples required to obtain emission from an entire landfill, possible interference with normal landfilling activities
Mass balance	few ha	continuous	CH ₄ and CO ₂	moderate	Few	Well suited for automation
Micro-meteorology	few ha	continuous	CH ₄ and CO ₂	moderate	Few	Demonstrated not to be applicable
Plume measurement	Entire landfill	hour	CH ₄	High	Some	Considered most accurate
Isotope measurement	Entire landfill	hour	CH ₄	very high	Some	Intended to measure amount of oxidation

At present there is no preferred methodology to obtain annual average emissions from an entire landfill site. Comparisons of methods by Savanne *et al.* (1995) and Scharff *et al.* (2003) conclude that no single method can measure both spatial and temporal variability of emissions, cost effectively, from a landfill surface (IPCC, 2006).

Estimating landfill gas emissions using models

The amount of CH₄ emitted from the surface of the landfill requires an estimate of the CH₄ potential of the waste (Peer and Thornloe, 1993). The development of LFG models started in the 1970's (Cossu *et al.*, 1996). Several methods have been used to estimate LFG emissions from landfills. Generally, simplified deterministic models are used. These models require knowledge of mechanisms controlling LFG production and describe these factors with simplified mathematical equations (Cossu *et al.*, 1996). A model chosen for LFG analysis should be based on the availability of data and the output required.

Models can be run for estimates from individual landfills and to attain national LFG estimates. Global estimates are obtained by summing national estimates (Morris, 2001). The most common types of models used to estimate LFG emissions from individual landfills are based on single-phase or multi-phase first order kinetics. These describe the decomposition of degradable organic fractions in waste and the resulting CH₄ generation (Cossu *et al.*, 1996).

First order models described by equation 1.9 assume waste to have a known half-life ($t_{1/2}$) where waste degradation occurs exponentially in time.

$$\alpha_t = \xi 1.87 A C_o k_1 e^{-k_1 t} \quad (1.9)$$

Where α_t represents the LFG production at a given time (m^3/year), ξ is the dissimilation factor (default is 0.58), A is the amount of waste in the landfill (Mg), C_o is the amount of organic carbon in the waste (kg C.Mg^{-1}), k_1 is the degradation constant (default 0.094) (y^{-1}) and t is the elapsed since waste deposition (y) (Cossu *et al.*, 1996; Scharff and Jacobs, 2006).

The majority of LFG estimate models follow first order kinetics (Cossu *et al.*, 1996) where the amount of product is always proportional to the amount of reactive material (IPCC, 2006). CH_4 generation is influenced by the amount of waste remaining or the amount of biogas already produced, and is not dependent on the waste deposited in the year of study but rather on the total mass of decomposing material in the landfill site at the time. This method assumes that the DOC in waste decays slowly over a few decades producing CH_4 and CO_2 . If conditions are constant, the rate of CH_4 production depends only on the amount of carbon remaining in the waste. Therefore, CH_4 emissions will be highest in the first few years after deposition, and then gradually decline as the degradable carbon in the waste is consumed by the bacteria responsible for the decay (IPCC, 2006).

In a multi-phase first order model waste fractions are divided into slow (wood and wood products), moderate (paper and textiles) or fast (food and garden) decomposing materials (EMCON, 1980; Cossu *et al.*, 1996; IPCC, 2006). Although many factors influence LFG production there is general agreement that first order kinetics with respect to substrate is suitable (Cossu *et al.*, 1996). Due to variations

in waste types, disposal rates, climate and operational conditions, the rate of LFG generation varies from landfill to landfill. Most LFG estimate models can account for this variability (Pitchel, 2005).

There are a number of key variables needed as input into the models including the quantity of waste placed in landfills annually and the composition of the waste. CH₄ production increases when waste deposited in the landfill has a high percentage of DOC, for example food and garden waste, paper and textiles. Site management and local or regional climate conditions also affect the rate of LFG production (Coops et al, 1995). It is important that data are available on all these factors before an attempt is made to model LFG emissions. The model output is closely associated with the input parameters values (Cossu *et al.*, 1996).

Early first order models used for estimates of LFG include the Palos Verdes Model, the Sheldon-Arleta Model and the Scholl Canyon Model. These were used on USA landfills in early 1980's (EMCON, 1980). Later models including LandGEM and GasSim are based on these older models.

A study by Scharff and Jacobs (2006) used six models to estimate landfill methane emissions from three landfills in the Netherlands. The models used were TNO (first order model) (Oonk and Boom, 1995), Afvalzorg (multi-phase model), LandGEM (USEPA, 2001), GasSim (Environment Agency UK and Golder Associates) (Gregory *et al.*, 2003), and EPER models France and Germany (Scharff and Jacobs, 2006).

TNO requires a limited number of parameters that consider the effect of depletion of carbon in waste. The carbon content is considered to be in a single phase, with all carbon present (decaying exponentially with time) being converted to LFG formation. For the Afvalzorg model, eight waste categories were divided into three fractions and LFG production is modelled for each fraction (Scharff and Jacobs, 2006). LandGEM, the USEPA model, determines the mass of CH₄ generated using the CH₄ generation capacity and the mass of annual waste deposited at the landfill since its opening. GasSim uses both a multi-phase model (described by van Zanten and Scheepers (1994)) and another approach based on the LandGEM model to calculate an estimate of CH₄ emissions (Gregory *et al.*, 2003). The multi-phase model utilises waste input in various categories and fractions and the detailed decomposition of the waste during the particular year of disposal. In the study average k values and a default oxidation factor of 10% were used in the multi-phase model and the AP-42 default values were used in the LandGEM approach.

The main difference between the GasSim model and the TNO, Afvalzorg and LandGEM is that GasSim allows for the inclusion of the extraction efficiency of the recovery system and the model then calculates the total CH₄ emissions. However, this element in the model only functions if the waste is capped to a certain degree (Scharff and Jacobs, 2006). LandGEM and GasSim are discussed in much detail in Chapter 2.

The modelled estimates of CH₄ emissions from the study were compared with measured results. It was concluded that the TNO, GasSim and LandGEM models overestimated CH₄ emissions from all three landfills whilst the Afvalzorg multi-

phase, German and French EPER models underestimated the emissions. There was a huge variation in results with the highest estimates obtained being 5 – 7 times higher than the lowest estimates. Lower variations in results were reported between the various measurement methods than between the different models, hence the need to integrate field measurements in models (Scharff and Jacobs, 2006).

The IPCC has reviewed a number of equations that allow total LFG to be estimated. These require a number of accurate country specific variables (Boeckx *et al.*, 1996; Bogner and Matthews, 2003). CH₄ emissions can be calculated from:

$$\text{CH}_4 \text{ emitted} = \{(\text{MSW}_t)(\text{MSW}_f)(\text{MC}_f)(\text{DOC})(\text{DOC}_f)(F)(16/12)-R\}(1-\text{OX}) \quad (1.10)$$

Where CH₄ emitted is in Tg/year, MSW_t is the total waste generated, MSW_f is the fraction of MSW disposed in the landfills, MC_f is the CH₄ landfilled fraction of MSW decomposing anaerobically, DOC is the fraction of degradable organic carbon, DOC_f is the fraction of DOC converted to CH₄ and CO₂, F is the CH₄ fraction of LFG (default is 0.5), R is the quantity of CH₄ recovered and OX is the CH₄ oxidation factor (default is 0) (Bogner and Matthew, 2003). Based on this equation the IPCC model calculates CH₄ emissions (Bogner and Matthews, 2003).

CALMIM (CALifornia Landfill Methane Inventory Model) estimates methane emissions using soil profile properties and microclimate modelling, differing from first order kinetic models (Spokas *et al.*, 2011; Bogner *et al.*, 2013; Bogner *et al.*, 2014). The LandGEM, GasSim, IPCC and CALMIM models are discussed in more detail in Chapter 2.

Landfill gas management systems: gas collection and utilisation

A number of technologies are available to manage and mitigate CH₄ emissions from landfills (Bogner et al, 2007) these can either be passive systems that vent gas to the atmosphere or active systems that mechanically extract LFG for utilisation (Pitchel, 2005). The cost of installing, operating and maintaining an active gas extraction system is much higher when compared to a passive system. Despite this, there are many benefits to actively extracting LFG (Pitchel, 2005).

Engineered landfill gas extraction systems include a series of horizontal and vertical wells that are installed within a cell at a landfill site. These wells are perforated and allow gas to be extracted using a pressurised system. Once an extraction system is in place the gas collected can then be flared, treated or used as an energy source for industrial boilers, heating and electricity generation (Pitchel, 2005; Stegmann, 1996). Most LFG extraction projects around the world utilise the gas to generate electricity by means of internal combustion or gas turbine engines (Stratchan et al., 2004). The first commercial project to actively extract and utilise landfill CH₄ was the Palos Verdes landfill site (California) in 1975 (Spokas et al., 2006). According to the United States Environmental Protection Agency (USEPA) approximately 1 million tonnes of MSW disposed at a landfill site can generate in the region of 8.5 m³.min⁻¹ of LFG which equates to roughly the production of 7 000 000 kWh of energy per year, enough to power 700 homes (Pitchel, 2005).

Extraction systems control the rate of LFG emissions from the surface of a landfill site to the atmosphere (Bogner *et al.*, 1995; Mosher *et al.*, 1999; Christopherson *et*

al., 2000; Christopherson and Kjeldsen, 2001; Spokas *et al.*, 2006) thus controlling the emissions of odours to the environment as sulphur compounds, volatile fatty acids and volatile amines are also extracted as LFG (DME, 2004). LFG utilisation estimates show that more than 5 TgCH₄/year can be reduced to atmosphere through LFG extraction systems (Hansen *et al.*, 1998; Willumsen, 2003; Bogner and Matthews, 2003). In 2004, LFG extraction projects were the second largest contributor to CH₄ reductions worldwide (Bogner and Lee, 2004). Extracting and utilising LFG also aids in reducing the migration and explosion hazard of CH₄ and in the extraction of leachate and condensate from the landfill sites (DME, 2004).

Previous studies in South Africa have shown that the warm climate and high precipitation rates at coastal areas in the country can encourage high rates of CH₄ generation, whilst seasonally dry cover soils can increase LFG emission rates to the surface (Fourie and Morris, 2004). Studies at specific landfill sites need to be conducted to better understand the relation between CH₄ generation, emissions, and oxidation in cover soils under South African climatic conditions (Fourie and Morris, 2004).

Lombard and Associates conducted a feasibility study for the Department of Energy (DME) which focused on testing the feasibility of using landfill sites for LFG extraction and utilisation for electricity generation in South Africa. Of the 453 landfills used in the study only 57 accepted enough waste volumes to justify being considered as a potential LFG to electricity project (DME, 2004). CO₂ emitted by producing electricity from LFG is much less than those emitted from fossil fuels and coal fired power stations thus making LFG a “greener fuel” (USEPA, 1991).

Currently there are only ten operational LFG extraction projects in South Africa. Of these only three generate electricity. These include:

- Two landfill sites within the eThekweni Municipality (Marianhill and Bisasar Road, generating 1MW and 6.5MW of electricity respectively).
- Four landfill sites within the Ekurhuleni Metro Municipality (at the Weltevreden, Rooikraal, Rietfontein and Simmer and Jack landfill sites). Currently only 1 MW of electricity is generated and exported directly to the municipalities grid at the Simmer and Jack landfill site.
- Alton landfill site in Empangeni generating 400kW of electricity. This is a privately owned landfill site.
- The privately owned Chloorkop landfill site within the Ekurhuleni Metro Municipality.
- Two landfill sites within the City of Johannesburg (Robinson Deep and Marie Louise landfill sites).

The New England Road landfill site in Pietermaritzburg has also registered a CDM project but only pumping trials have been conducted at the site and to date it has not been commercialised. Due to the fact that LFG extraction and utilisation is not a legislative requirement in South Africa it places these projects in a distinctive position to be exploited under the Clean Development Mechanism (CDM) as it meets the requirement of additionality (Bogner and Lee, 2004). Additionality which means “... reductions in emissions that are real, measurable and additional to any that would occur in the absence of the certified project activity” (Bogner and Lee, 2004). All ten LFG extraction projects active in South Africa are registered CDM projects. The success of these projects on the carbon credit market depends on how

attractive they are to international investors (Bogner *et al.*, 2004). To gauge the potential for other LFG extraction projects to enter the carbon market there is a need to accurately quantify and estimate LFG emissions at landfills in South Africa so as to identify suitable landfill sites that can be viable for gas extraction systems.

Kyoto protocol and Clean Development Mechanism

The Kyoto protocol stems from the United Nations Framework Convention on Climate Change (UNFCCC) which was formed after negotiations on a global climate protection treaty led to its establishment at the Earth Summit in 1992 and signed into effect in March 1994 (Fenhann and Hinostroza, 2011). The main objective of the UNFCCC is to stabilise greenhouse gas concentrations in the atmosphere to reduce the effect of climate change. It divides countries into Annex I Parties (developed or industrialised countries including members of the OECD (Organisation for Economic Co-operation and Development) and economies in transition (the EIT Parties)), Annex II countries (consist of the OECD members of Annex I, but not the EIT Parties) and non-Annex I Parties (developing countries).

In addition, 49 Parties are classified as least developed countries (LDCs) by the United Nations and are given special consideration under the Convention. The Kyoto protocol of the Convention sets legally binding emission reductions to Annex I parties to reduce their GHG emissions between 2008 and 2012 also referred to as the first commitment period (CP1), by approximately 5% (on average as each Party has an “Assigned Amount” for reduction) below their GHG concentrations in 1990 (Kyoto Protocol, 1997; Fenhann and Hinostroza, 2011). In the second commitment

period from 2013 to 2020 (CP2) Parties (differing from the CP1 Parties) committed to reduce GHG emissions by at least 18 % below 1990 levels from 2013 to 2020 (www.unfccc.int).

The Kyoto Protocol has three flexible mechanisms to assist Annex I Parties in reducing their emission reductions. This includes:

- International Emissions Trading which allows Annex I Parties to sell their excess Assigned Amount Units (AAU's) to other Annex I Parties that require them
- Joint Implementation allows Annex I Parties to sell emission reduction units (ERU's) from emission reduction projects between themselves
- Clean Development Mechanism (CDM) allows Annex I Parties to invest in emission reduction projects in non-Annex I Parties which produce Certified Emission Reductions (CER's) that are sellable on the carbon market. These projects must also assist these countries in achieving their sustainable development goals

All CDM projects undergo a project cycle that begins with the development of a Project Design Document (PDD) which is approved by the host country's Designated National Authority (DNA) (Figure 1.7). Once approval is received which according to the South African DNA should occur within 45 days from submission of the PDD, the project is validated by a Designated National Authority (DOE) and then registered as a CDM project by the UNFCCC Executive Board (EB). The project should show "additionality" that is, the emission reductions associated with the project would not occur without the existence of the project.

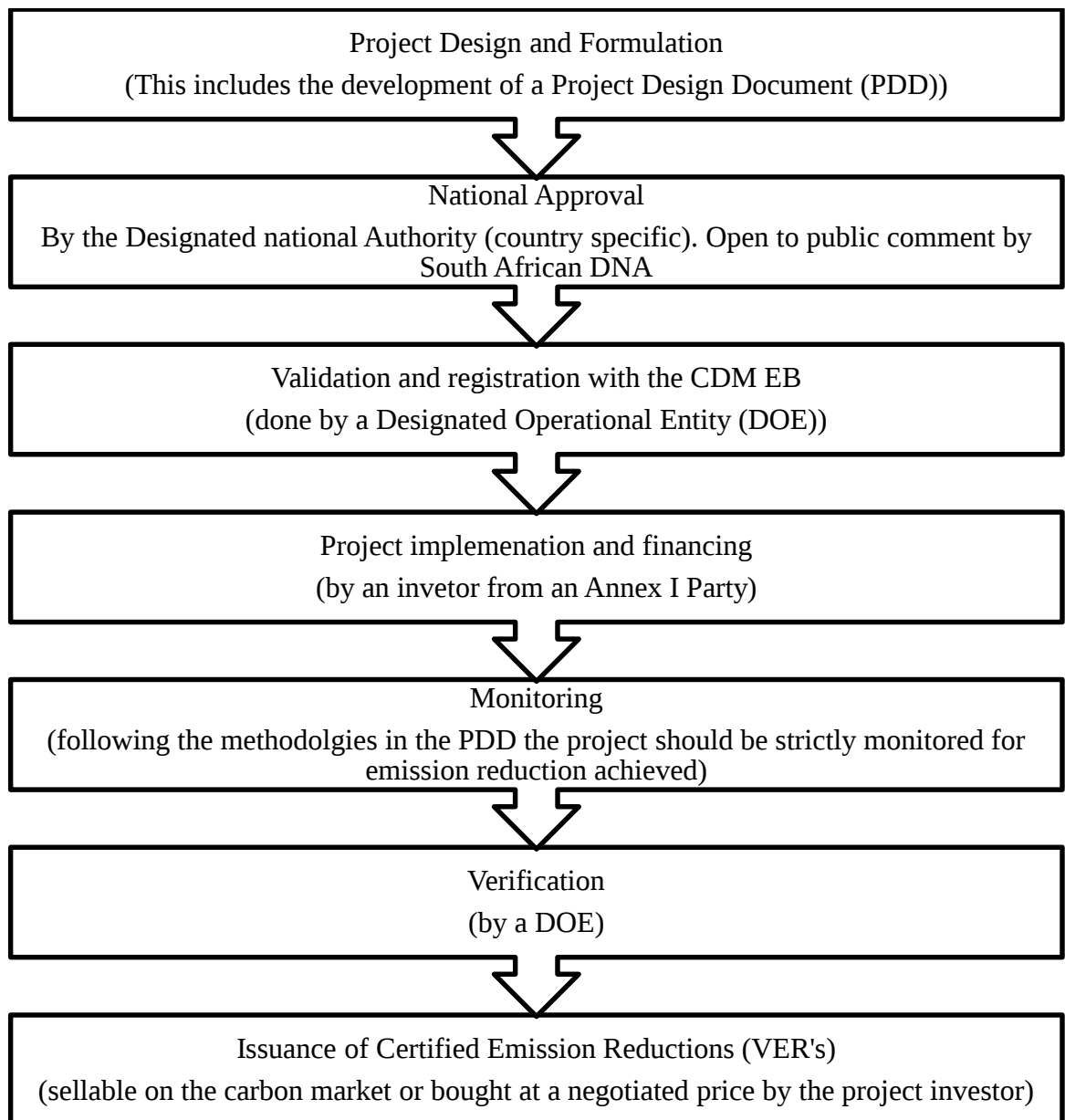


Figure 1.7: CDM project cycle (after UNEP collaborating Centre on Energy and Environment, 2002 and South African DNA project cycle (www.energy.gov.za)).

After registration financing and project implementation takes place in which continuous monitoring of the emission reductions occur in strict accordance with the PDD and approved monitoring methodology.

A monitoring period can then be selected where the emission reduction undergo a verification process to ensure they are real and comply with CDM regulations after which they are issued as CER's which can be sold on the carbon market. This project cycle takes several years to complete and requires a large amount of financing through capital costs as well as operational and maintenance costs throughout the life of the project.

The European Emission Trading Scheme (EU ETS) is the largest emission reductions trading scheme in the world followed by China. The EU ETS reduced emissions by 2 084 Mt/CO₂eq in 2013 whilst China recovered 1 115 Mt/CO₂eq (World Bank, 2014). In 2011 only 2% of CDM projects are registered with the UNFCCC. If South Africa and countries in North Africa are not considered this percentage drops substantially to just 0.6% (ISS, 2011).

Of the 358 CDM projects submitted to the South African DNA (which includes bio-fuels, energy efficiency, waste management, cogeneration, fuel switching and hydro-power projects) only 86 have been registered by the CDM Executive Board of which 12 have been issued with CER's (www.energy.gov.za). 43.8% of projects are classified as renewable energy with a potential emission reductions of 44.8 Mt/CO₂eq whilst methane recovery and flaring projects only account for 5% of projects contributing to 6.09 Mt/CO₂eq in emission reductions (Figure 1.8). These are inclusive of projects that have been submitted and awaiting approval for registration.

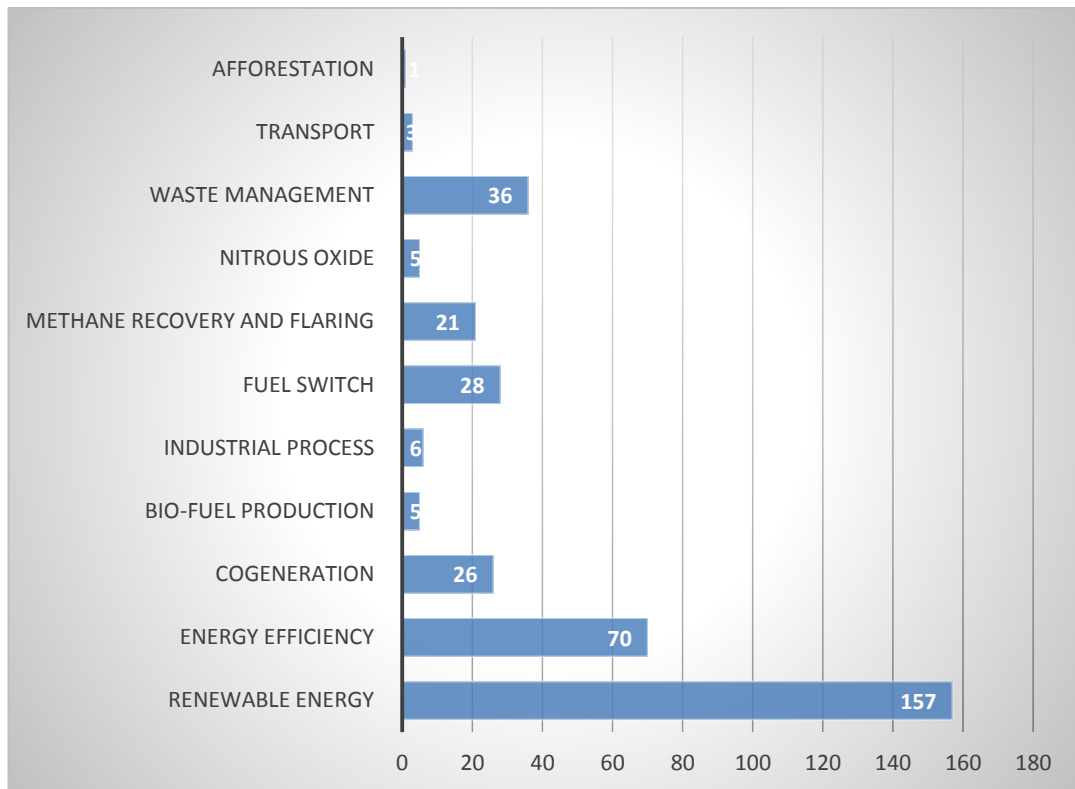


Figure 1.8: CDM projects portfolio in South Africa as of March 2015 (after www.energy.gov.za).

Whether or not CDM truly contributes to sustainable development is debated. There is a consensus in the literature that the CDM places more emphasis on emission reductions at the disadvantage to poverty alleviation, social and economic development of host countries (Brown et al. 2004; Ellis et al. 2004; Cosbey et al. 2005; Pearson 2005; Michaelowa, 2007; Olsen 2007; Schneider, 2007; Sutter and Parreno, 2007) highlighting that it does not provide funding to renewable energy projects that have high “co-benefits” (projects contribution to job creation, environmental protection and other sustainable development initiatives).

Sutter and Parreno (2007) found that only 1.6 percent of carbon credits were from projects that benefited sustainable development. However, the CDM does work

sufficiently well in its contribution to low cost emission reduction projects and investments into cleaner technologies. Thus, turning its contribution to sustainable development around by offering a higher price for CER's from projects with a strong contribution to sustainable development can increase the investment into such projects (Sutter and Parreno, 2007). The SD-TOOL 01 "Voluntary tool for describing sustainable development co-benefits of CDM project activities or programmes of activities (POA)" developed by the UNFCCC in 2014, is a voluntary tool that project owners can use to calculate their contribution to sustainable development.

A Sustainable Development Co-benefits (SDC) description report is produced and placed on the associated projects CDM webpage (at the UNFCCC) which allows potential buyers of credits to gauge what the contribution to sustainable development is. In addition, the EU ETS took a decision that parties may exchange their excess CER's from CP1 for allowances valid after 2012 if the CDM project was registered during CP1 or with CDM projects registered post 2012 (CP2) only in Least Developed Countries. This decision should then strengthen the sustainable development leg of the CDM by allowing Annex I Parties to invest in CDM projects in LDC.

Despite the carbon market being a multi-billion dollar industry, the deteriorating economic situation since 2010 as well as the large quantity of CER's available on the carbon market meant that investments in CDM projects were not a priority leading to the rapid decrease in the value of CER's. This has been the major stumbling block to the success of the CDM. The current price of CER's in the EU

market are between US \$0.30 and \$0.60 (World Bank, 2014) cents which is very discouraging to the development of future CDM projects. It means that the capital, operational and maintenance costs of implementing a CDM projects does not balance or create a return on investment solely from the sales of the CER's. In 2013 there was an 88% decrease in CDM projects submitted for validation and ten times fewer projects registered in 2013 compared to 2012. In addition, over the past three years March 2014 had the lowest monthly CER issuances (World Bank, 2014).

With no sign of a short-term recovery in demand of carbon credits the situation is exacerbated by the withdrawal of Japan, New Zealand and Russia from CP2 of the Kyoto Protocol as well as the reduction in the number of financial institutions, private sector bodies and DOE's currently active in the carbon credit space (World Bank, 2014). Despite this, globally at domestic level there is an increased focus on climate change policy and GHG mitigation actions are at the forefront of development. Figure 1.9 represents the distribution of GHG emissions globally. Based on the significant difference in GHG emissions between countries there is a strong need for international collaboration in emission reductions (World, bank, 2014). Many countries are introducing different methods to mitigate their GHG emissions including domestic cap and trade initiatives, carbon taxation, emission trading schemes and carbon offsets programmes.

South Africa's carbon tax combined with an offsets component will come into effect in January 2017. The National Treasury released the draft Carbon Offsets Paper for comment on 29 April 2014 and the Draft Carbon Tax Bill in November 2015. The carbon tax will cover GHG emissions from fuel combustion and

industrial process emissions, covering 80% of the total GHG emissions in the country (World Bank, 2014). The inclusion of an offsets component in the proposed carbon tax regime in South Africa will provide flexibility for entities to comply with the requirements of the tax and is likely to encourage voluntary emission reductions. The carbon offsets component will accept carbon credits from projects registered under existing, internationally recognized, carbon standards including CDM. The intended price per t/CO₂ eq is set at R120 (US \$12 t/CO₂ eq) at the start of the carbon tax and increasing by 10% until 2019 (the transitional period of the tax).

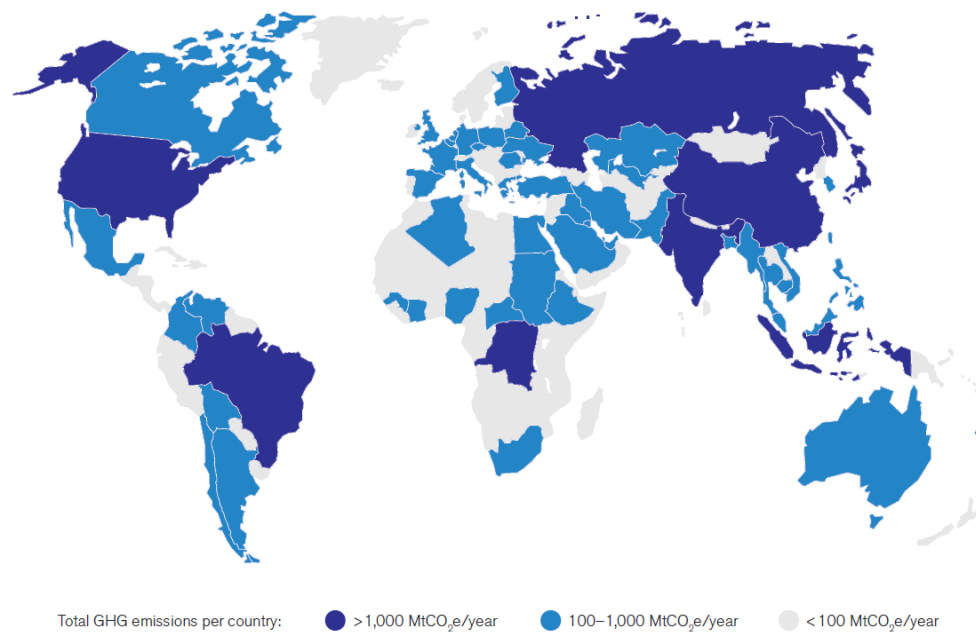


Figure 1.9: GHG emissions globally per country (World Bank, 2014).

This value is significantly higher than the current pricing of CER's on the EU carbon market which will assist many registered CDM projects in the country that are currently at risk of shutting down.

For waste management projects in South Africa there is a strong move forward to support sustainable product design, resource efficiency and waste prevention focusing on waste minimization and energy recovery. This research aims to evaluate the contribution of landfill gas emissions to the GHG emissions in South Africa and what the impact of the CDM has had on the implementation and success of landfill gas extraction and utilisation projects thus mapping a road for the development of larger scale Waste to Energy facilities to encourage the waste hierarchy and promoting sustainable development in the country.

Research aims and hypotheses

CH₄ emissions monitoring needs to be conducted to better understand the relative dynamics of CH₄ generation, emissions and oxidation under South African climatic conditions. This will aid in assessing the associated contribution of landfills to anthropogenic CH₄ emissions in South Africa. Other gases such as H₂S and volatile organic compounds also need to be monitored to assess the influence on local air quality and the development of odour nuisances. In addition, subsurface gas concentrations at the boundary of the landfill should be monitored to determine lateral migration of LFG from the landfill site into the surrounding environment which could create an explosion hazard. The main objectives of this research are as follows:

1. to predict LFG generation from landfills in South Africa using theoretical models and compare the results to actual measured LFG yields.
2. to measure CH₄ and CO₂ fluxes from the landfill surface.

3. to assess the impact of LFG emissions on the air quality of the surrounding environment.
4. to assess the feasibility and functionality of gas extraction and utilisation at landfill sites in South Africa to mitigate greenhouse gas emissions.

The LFG generated at landfill sites can be estimated on the basis of the amounts and composition of the waste disposed as well as the waste management practices imposed. All landfills used in this study are managed municipal solid waste disposal sites and are administered by guidelines set out in the DWAF Minimum Requirements for Waste Disposal by Landfills as well as the National Environmental Management: Waste Act, 2008 (Act 59 of 2008) and its accompanying Waste Regulations. Research has shown that waste composition affects both the theoretical yield and rate of CH₄ production in anaerobic landfills. Meteorological conditions such as rainfall, temperature and atmospheric pressure changes also affect LFG production.

The key research questions to be addressed in this project are:

1. Do LFG emissions from landfill sites make a significant contribution to anthropogenic CH₄ emissions in South Africa?
2. Do landfill gas emissions significantly affect the air quality of the local environment around the landfill?
3. How successful is LFG extraction and utilisation in mitigating emissions from landfills?

Chapter 2

Data and methodology

Estimating and quantifying landfill gas emissions is important from an environmental, regulatory and financial perspective. The significant contribution of landfill CH₄ to global warming and the market for emission reductions places an increased emphasis on the need for accurate methods to calculate the generation and emissions potential of landfills. This chapter outlines the methodology used in this study to estimate and measure LFG emissions from landfills in South Africa. A description of each of the landfill sites selected for this study is included. Procedures for subsurface gas generation data collection and analyses are discussed. Detailed explanations on the LFG estimation models used in the study (LandGEM, GasSim, the IPCC Waste Model 2006 and CALMIM) are provided with a focus on input data used for the models. The methodology used for surface monitoring of pollutants and dispersion modelling are discussed in detail. Methodologies used for LFG extraction and utilisation or flaring are also discussed using the Ekurhuleni Metro Municipality Clean Development Mechanism (CDM) project as a case study.

Description and locations of study sites

Nine landfills in Gauteng, South Africa were used to estimate landfill gas emissions in this study – five located in the City of Johannesburg (CoJ) and four within the Ekurhuleni Metro Municipality (EMM) (Figure 2.1). Their classification, available airspace and lifespan and size are listed in Table 2.1. The landfills range in size and age (year of establishment). This variation enables a good comparison with respect to waste streams and moisture content and availability and the resultant LFG emission predictions. Municipal solid waste (MSW) including domestic waste, builders rubble, garden waste and light industrial waste with the exclusion of

medical and hazardous waste, are accepted at the sites for disposal. Co-disposal of general waste and sewage sludge takes place at Goudkoppies, and Marie Louise. The Rietfontein landfill sites accepts de-listed waste and sludges.

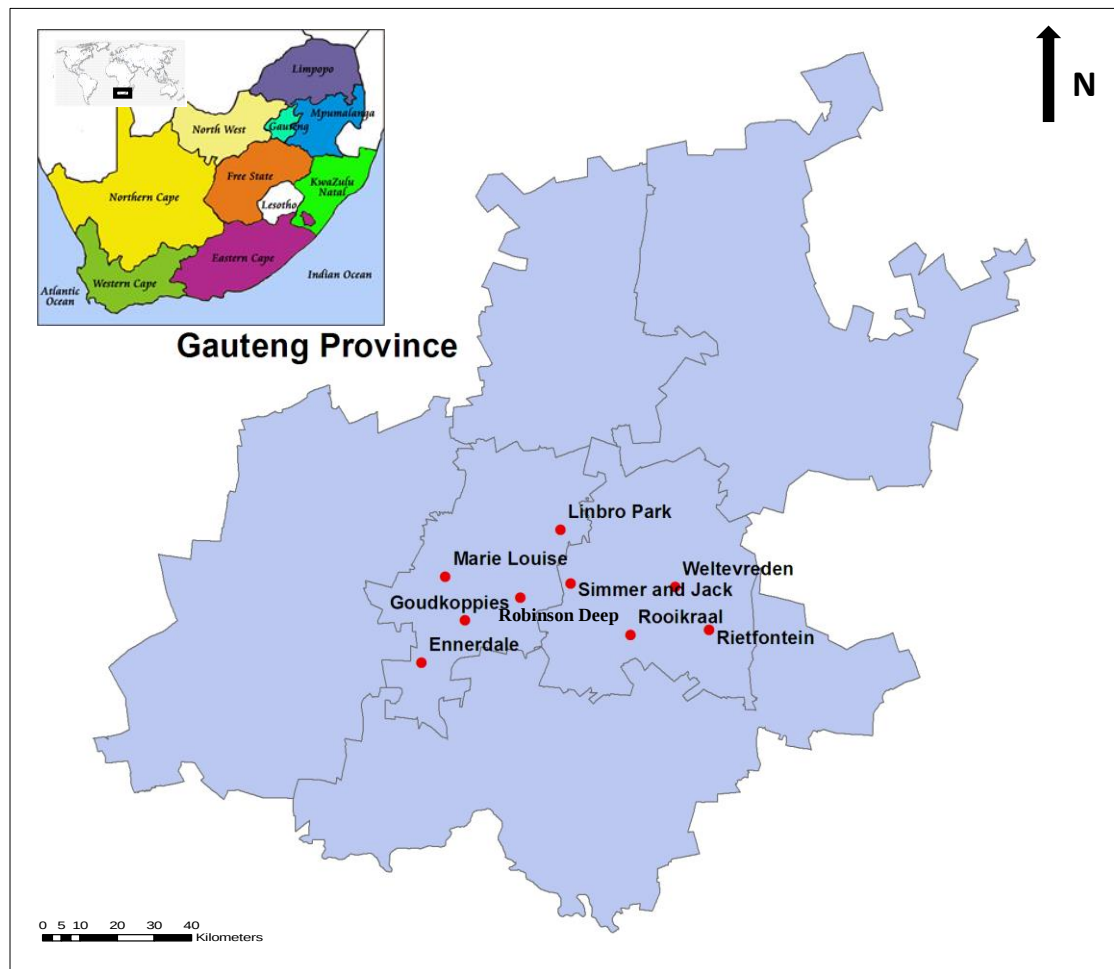


Figure 2.1: Location of landfills used in this study. Marie Louise and Robinson Deep service the greater part of the inner city of Johannesburg Metro whilst Ennerdale and Goudkoppies service the south and Linbro Park the north. Rietfontein and Weltevreden services the eastern areas of EMM, Rooikraal the south and Simmer and Jack the western parts of EMM.

City of Johannesburg landfill sites

The City of Johannesburg Local Municipality is the largest city in South Africa.

The municipality covers an area of 1 645 square kilometres and is divided into seven

regions consisting of 130 wards. According to the census conducted into 2011 the City has a total population of 4 434 827 and 95.3% of households have their waste removed on a weekly basis (www.stassa.go.za, accessed 3 April 2015). Pikitup, an entity operated by the CoJ Municipality is responsible for the collection and disposal of waste within the City. MSW is disposed at 5 landfill sites as described below.

Goudkoppies is located approximately 20 km to the south west of Johannesburg and covers 57.2 hectares of land. The landfill accepts approximately 20 000 tonnes of waste a month and is divided into three phases. It is a Class B (GLB⁻) landfill site without any significant leachate production. A new section was added to the current site in 2004 that will increase the site lifespan by 50 years. The new section is undeveloped and not considered in the modelling for this study.

Marie Louise is located approximately 20 km to the west of Johannesburg and services the western part of the Greater Johannesburg Metro and the northern parts of Soweto. The landfill covers 44 hectares and the surrounding land is mainly used for industrial and mining purposes. The landfill accepts approximately 30 000 tonnes of waste a month and is classified as a Class B (GLB⁺) with significant leachate production. The landfill is divided into seven phases. Phase one has reached its final landform height and is no longer operational. The area has been capped with a 300mm clay layer.

Robinson Deep is located approximately 3 km south of Johannesburg. The landfill services the western part of the Greater Johannesburg Metro and the northern parts

of Soweto. The landfill was commissioned in 1930 and is a Class B (GLB⁻) site. The landfill accepts waste from the southern suburbs and the Johannesburg CBD. It collects industrial and residential waste. The site was built upon layers of shale, clay and sandstone, over an old slimes dam. The surrounding areas are mostly residential and industrial. The Turffontein race course is also situated adjacent to the landfill.

Ennerdale is located approximately 32 km south west of Johannesburg. The landfill consists of five phases. Approximately 10 000 tonnes of waste are accepted at the site per month. The total extent of the site is 16.3 hectares. The general slope direction of the landfill is from south to north-east. Approximately 10 000 tonnes of waste are accepted at the site per month and is classified as a Class B (GMB⁻). It is a medium sized landfill and situated adjacent to industry (brickworks) and farmland.

Linbro Park is located approximately 15 km north east of Johannesburg. The landfill consists of five phases. The total extent of the site is 84 hectares. The landfill is not operational and was closed in September 2006. Approximately 800 tonnes of waste was accepted at the site on a daily basis. The landfill has been rehabilitated with a final cover.

Ekurhuleni Metro Municipality landfill sites

The EMM has a population of approximately 3 178 473 (according to census 2011) covering 1 975 square kilometres. The City is highly urbanised with a large number of industrial activities taking place, as it is home to South Africa's largest airport,

OR Tambo International Airport. It has a population growth rate of 2.47% (www.stassa.go.za, accessed 3 April 2015). EMM is divided into 101 wards with major suburbs and townships being Germiston, Tembisa, Kempton Park, Bedfordview, Benoni, Boksburg, Springs, Vosloorus and Thokoza. The EMM owns and operates five landfill sites. Current waste disposal is approximately 1.65 million tonnes per annum. All sites are well engineered with leachate and storm water collection systems. Landfill gas extraction takes place at four sites, which are used in this study and described below.

Rooikraal landfill is located within Boksburg and opened in 1988. It operates under permit number 16/2/7/C221/D24/Z1/P512 and covers 98.4 hectares of land. It consists of four phases. Stage 1 is unlined, has reached capacity and is covered with a 500 mm soil layer. Stage 2A and Stage 2B are currently operational. Stage 3 has been recently developed and is not yet operational and Stage 4 is undeveloped. The site accepts approximately 20 000 – 30 000 tonnes of waste on a monthly basis and has roughly 4.9 million tons of waste in place. It is classified as a Class B (GLB⁻) site. A 2 000 m³/hr flare is installed at the site. Thirty six vertical wells are installed in Stage 1 and fifteen horizontal wells are installed in Stage 2A and Stage 2B.

Simmer and Jack landfill is a Class B (GLB⁻) site, situated in Germiston and has been operating since 1983 under permit number B33/2/0322/494/P223. The site has almost reached capacity and is anticipated to reach closure in 2016. The landfill covers 60 hectares and is divided into seven phases. Cells one to six have reached capacity, Cell one has a cap of 500 mm soil and Cell 2-4 has a 300 mm soil cap. The landfill accepts between 15 000 and 20 000 tonnes of waste on a monthly basis

and has roughly 2.5 million tonnes of waste in place. A 3 000 m³/hr flare is installed at the site. Thirty eight vertical wells are installed in Cell 2, 3 and 4. Twelve vertical wells and ten horizontal wells are installed in Cell 5 and 6.

Weltevreden landfill situated in Brakpan has been operating since 1994. It is a Class B (GLB⁻) site and is anticipated to close in 2039. The site operates under permit number B33/2/321/172/P137 and covers 128 hectares of land. It accepts approximately between 20 000 and 25 000 tonnes of waste a month and has more than 2.1 million tonnes of waste in place. The landfill has a large area that is still to be developed. There are seven developed phases. Phases one to five have reached capacity. Cell 1 has a 500 mm soil cap. Cell 6 and 7 are currently operational. A 2 000 m³/hr flare is installed at the site. There are eighteen vertical wells installed in Cell 1, four vertical wells and four horizontal wells in Cell 2 and 3 and seventeen horizontal wells installed in Cell 4 and 5.

Rietfontein landfill site is operational since 1997, covers 40 hectares of land and is situated in Springs. The site accepts approximately 20 000 tonnes of waste a month. It is a Class B (GLB⁺) site operating under permit 16/2/7/C221/D494/P275 and accepts de-listed sludges and liquids. The site has approximately 1.1 million tonnes of waste in place and closure is anticipated in 2023. The site is divided into Area A and B and consists of five phases. Area A consists of Cell 1 to 3 and has reached final landform height and has a final capping of 500 mm clay. There are 25 vertical wells installed in Cell 1, 2 and 3 and a 1 300 m³/hr flare. Area B is currently operational and consists of Cell 4 and 5. No gas wells are installed in Area B.

Table 2.1: Characteristics of landfills used in this study.

Landfill site	LF G reco very	Opening year of landfill (year)	Landfill classification	Types of waste handled						Landfill capacity (m ³)	Monthly waste acceptance rates (tonnes)	Landfill Size (hectares)	Site location
				Domestic	Industrial	Garden waste	Ash	Building rubble	Sludge/Liquid				
Goudkoppies		1989	operational GLB ⁻	X	X	X	X	X	X	12.0 x 10 ⁶	19000 – 25000	57.2	26°16'50.39"S 27°55'25.90"E
Marie Louise	X	1991	operational GLB ⁺	X	X	X	X	X		8.5 x 10 ⁶	28000 – 35000	44	26°11'25.83"S 27°52'59.80"E
Robinson Deep	X	1997	operational GLB ⁻	X	X	X	X	X		24.8 x 10 ⁶	15000 – 17000	124	26°14'1.35"S 28°2'16.36"E
Ennerdale		1988	operational GMB ⁻	X	X	X	X	X		1.7 x 10 ⁶	4000 – 10000	16.3	26°22'5.31"S 27°50'0.41"E
Linbro Park		1989	closed GLB ⁺	X	X	X	X	X	X	12.0 x 10 ⁶	19000- 25000	57.2	26° 5'37.46"S 28° 7'11.40"E
Rooikraal	X	1988	operational GLB ⁻	X	X	X	X	X		19.6 x 10 ⁶	20000 -25000	98.4	26°18'40.21"S 28°15'52.07"E
Simmer and Jack	X	1983	operational GLB ⁻	X	X	X	X	X		12.0 x 10 ⁶	15000-20000	60	26°12'16.27"S 28°8'28.48"E
Weltevreden	X	1994	operational GLB ⁻	X	X	X	X	X		25.6 x 10 ⁶	20000-25000	128	26°12'40.59"S 28°21'24.91"E
Rietfontein	X	1997	operational GLB ⁺	X	X	X	X	X	X	8.0 x 10 ⁶	20000-25000	40	26°17'59.81"S 28°25'34.48"E

Methodology

A number of methods were considered to meet the objectives of this study. LFG estimation models LandGEM, GasSim, the IPCC Waste Model 2006 and CALMIM were used to estimate emissions from the landfills described above. It is important to consider waste volumes and composition, local climate and other parameters that are required as input for the models. Model results are compared to previous studies globally to assess the suitability of the models to South African landfills. Actual gas yields from landfill gas extraction projects in EMM and CoJ will be compared to modelled results. Measured concentrations of subsurface gas generation are also analysed. Surface emissions were monitored using flux chambers at the surface of the landfill sites as well as using a mobile caravan to monitor ambient emissions. Source apportionment is applied to analyse the contribution of the landfill site to emissions.

Landfill gas estimation models used in study

The LFG estimation models LandGEM, GasSim and IPCC Waste Model 2006 were used in this study to estimate LFG production and emissions. A number of parameters were needed as input into the model including the design capacity of the landfill and the opening and closure year of the landfill. The most important parameters are the annual waste acceptance at the site, the methane generation rate (k) and the potential methane generation capacity (L_0) (Thornloe *et al.*, 1999). The models can be run using model default data or site specific data. All models were run using the same values for k , L_0 and the CH_4 and CO_2 fractions. Varying values

for different parameters were used whilst conducting a sensitivity analysis and testing model applicability to a semi-arid environment.

The Landfill Gas Estimation Model (LandGEM)

The Landfill Gas Estimation Model (LandGEM) was developed by the Clean Air Technology Centre (CTC) of the USEPA (Pelt *et al.*, 1998). The first version of the model was released in 1998. The version used in this study is version 3.02 released in May 2005 and replaces previous versions (2.01 and 3.01).

The model can be downloaded from the USEPA website <<http://www.epa.gov/ttn/catc/dir1/landgem-v302.xls>>. The model is classified as a simplified deterministic model according to Cossu *et al.*, (1996) as it uses a simplified mathematical equation to describe the decay of waste. It is based on a first order decomposition rate reaction. Model defaults are based on empirical data from US landfills (Alexander *et al.*, 2005). Inventory default values are based on emission factors in the US Environmental Protection Agency (USEPA) Compilation of Air Pollutant Factors (AP-42). The first order decomposition rate equation used by the model is as follows:

$$Q_{CH_4} = \sum_{i=1}^n \sum_{j=0.1}^1 kL_o \left(\frac{M_t}{10} \right) e^{-kt_j} \quad (2.1)$$

Where Q_{CH_4} is the annual CH₄ generation in the year of the calculation (m³/year), i is the one year time increment, n represents the year of calculation minus the initial year of waste acceptance, j is the 0.1 year increment, k is the methane generation

rate (year^{-1}), L_o is the potential methane generation capacity (m^3/Mg), M_i is the mass of waste accepted in the i^{th} year (Mg) and t_{ij} is the age of the j^{th} section of waste mass M_i accepted in the i^{th} year (decimal years).

The model requires various inputs, the most important of which are the waste acceptance rates since the opening of the landfill to the current year. It limits the number of years of waste acceptance data to 80 years.

The model calculates the landfill closure year based on the Landfill Open Year, Waste Design Capacity and Waste Acceptance Rates entered (Equation 2.2) by the sum of the Waste Acceptance Rates subtracted from the Waste Design Capacity divided by the current acceptance rate entered. The model assumes that the final waste acceptance rate entered will continue to be the annual amount of waste accepted until closure of the landfill (USEPA, 1998; Liphoto, 2001).

$$\text{Calculated landfill closure year} = \left[\frac{(\text{Waste design capacity} - \sum (\text{Waste acceptance rates}))}{\text{Final waste acceptance rate in current year}} \right] + \text{Year of final waste acceptance rate} \quad (2.2)$$

The CAA default values in the model provide emission estimates that reflect the maximum emissions and are generally used for regulatory purposes. To estimate actual emissions in the absence of site-specific data, a second set of default values (the inventory defaults) is provided in the model. The inventory default values in the model are based on emission factors from the U.S. Environmental Protection Agency's Compilation of Air Pollutant Emission Factors, AP-42 (USEPA, 1998).

The CH₄ generation rate constant, k , determines the rate of CH₄ generation from the landfill. The higher the value of k , the faster the CH₄ generation rate increases and then decays over time. The value of k is a function of waste moisture content, availability of the nutrients for methanogens, pH, and temperature. k is assumed to peak at closure of the landfill. The value for the potential CH₄ generation capacity of waste (L_o) depends on the type of waste in the landfill. The higher the cellulose content of the waste, the higher the value of L_o (USEPA, 1991).

GasSim

GasSim has been developed using the HELGA framework which was developed for the UK Environment Agency. The principal drivers behind the development of GasSim were firstly the need to assess the health risk for people living and working near landfills and secondly as a management tool for the reduction of greenhouse gas emissions from landfills in the UK. GasSim2 used in this study replaces the older versions GasSim 1.0 – 1.5 (Attenborough et al, 2005; GasSim2, 2005). It considers the uncertainty in input parameters using a Monte Carlo Simulation.

GasSim2 is a conceptual model that considers the landfill as individual cells, each with its own engineering and waste composition. The central conceptual model is a landfill containing biodegradable wastes, which generate bulk gases comprising mainly CH₄ and CO₂. It is divided into five modules namely:

- source term module
- emissions module
- atmospheric dispersion module

- lateral migration module and
- exposure/impact modules

Progression to the different modules is only necessary if the information is required. In this study only the source term, emissions and lateral migration modules will be used. GasSim estimates LFG emissions using two approaches. The first approach uses the GasSim multiphase equation, which is based upon a multi-phase model described by van Zanten and Scheepers (1994) and the second approach is based on the USEPA's LandGEM model (Scharff and Jacobs, 2006).

The source term module determines LFG generation using a first-order decay model and multi-phase equations, for an individual cell or entire landfill site based on the mass of waste deposited and its composition for up to 200 years (GasSim2, 2005). In addition the module requires input based on the landfill characteristics including infiltration rates and landfill cap and liner engineering. The degradation of organic material present in the waste and its conversion to LFG is represented by equation (2.3):

$$C_t = C_o - C_{o1}e^{(-k_1t)} + C_{o2}e^{(-k_2t)} + C_{o3}e^{(-k_3t)} \text{ and}$$

$$C_x = C_t - C_{t-1} \quad (2.3)$$

Where C_t is the mass of degradable carbon degraded up to time t (tonnes), C_o the mass of degradable carbon at time $t = 0$ (tonnes), $C_{o,i}$ represents the mass of degradable carbon at time $t = 0$ for each fraction of waste, C_x is the mass of carbon

degraded in year t (tonnes), t represents the time between waste placement and LFG generation (years) and k_i is the degradation constant for each fraction of degradable carbon (per year) (GasSim2, 2005).

The emission module uses the output from the source term module and calculates LFG emissions to the surface. The module does however consider CH_4 oxidation rates through cover soils and gas management systems (collection, flaring, utilisation) if present. Model default values (based on the IPCC guidance values) will be used for CH_4 oxidation rates of all landfills used in this study. In order to meet the objectives of this study to compare modelled emissions to actual gas generation at the landfill sites, no gas management systems were included in the model runs. The model can also assess the landfills impact on the global atmosphere by determining the global warming potential of the LFG emissions.

The lateral migration module simulates the transverse subsurface migration of LFG through advection and diffusion. The model simplifies the surrounding subsurface environment into one zone, simulated using a conventional one dimensional linear pathway. This provides the maximum concentration at a given point at any distance from the landfill and can be used to determine exposure to humans and vegetation including the migration of LFG into buildings (GasSim2, 2005).

Uncertainty in the model is present, firstly by the assumptions made in the equations used and secondly confidence and error on the site specific data used for input. The model runs using a Monte Carlo analysis. It allows for the use of PDF's (Probability Density Functions) that could reduce the uncertainty related to site data. PDF's

allow a range of values to be used as input for each parameter instead of a single number. Distributions include uniform, triangular, log uniform or log triangular and normal or log normal distribution. A single value can be entered if the data are known to be highly accurate and precise (GasSim2, 2005).

IPCC Waste Model 2006

The Revised 1996 IPCC Guidelines for National Greenhouse Gas Inventories and the Good Practice Guidance and Uncertainty Management in National Greenhouse Gas Inventories (IPCC, 2000) describes two methods for estimating CH₄ emissions from landfills, the mass balance method and the First Order Decay (FOD) method. In the 2006 Guidelines for National Greenhouse Gas Inventories the mass balance method is not considered. The current IPCC methodology is based on the FOD method (IPCC, 2006).

The IPCC Waste Model 2006 is a first order multi-phase model based on waste composition data. The amounts of degradable waste material (food, garden and park waste, paper and cardboard, wood, textiles) contained in the waste are entered separately. The model can be downloaded from the IPCC website http://www.ipccnggip.iges.or.jp/public/2006gl/pdf/5_Volume5/IPCC_Waste_Model.xls. A choice of three tiers, depending on the data available, can be used to estimate the CH₄ emissions. Tier 1 is based on the FOD method using mainly default activity data and default parameters. Tier 2 uses the FOD method and some default parameters, but requires good quality country-specific activity data on current and historical waste disposal. Historical waste disposal data for 10 years or

more should be based on country specific statistics, surveys or other similar sources. Tier 3 is based on the use of good quality country-specific activity data and the use of either the FOD method with nationally developed parameters or measured country specific parameters (IPCC, 2006).

A key input in the model is the amount of DOC in waste disposed at landfill sites (DOC_m), half-life ($t_{1/2}$), L_o and the fraction of DOC which decomposes (DOC_f). Data on the different waste types (food, paper, wood, textiles) is needed or alternatively as mean DOC in bulk waste disposed (IPCC, 2006). DOC_f is an estimate of the fraction of carbon that is ultimately degraded and released from the landfill and considers the fact that some DOC does not degrade or degrades very slowly under anaerobic conditions. Wood and wood products decay very slowly and accumulate in the landfill (long-term storage). Carbon fractions in other waste types decay over varying time periods. The long term storage of carbon in paper and cardboard, wood, garden and park waste is of special interest as the changes in carbon stock in waste originating from harvested wood products. Assuming that the landfill is in the anaerobic stage and the DOC values include lignin, the recommended default value for DOC_f is 0.5 (Oonk and Boom, 1995; Bogner and Matthews, 2003). Tier 2 and Tier 3 methodologies allow the user to enter separate DOC_f values for specific waste types.

Decomposable Degradable Organic Carbon ($DDOC_m$) is the organic carbon that will degrade under the anaerobic conditions in the landfill. $DDOC_m$, described in Equation 2.4 equals the product of the waste amount (W), the fraction of degradable organic carbon in the waste (DOC), the fraction of the degradable organic carbon

that decomposes under anaerobic conditions (DOC_f), the methane correction factor (MCF) which accounts for waste that decomposes under aerobic conditions (IPCC, 2006). L_o (Equation 2.5) is calculated using $DDOC_m$, the CH_4 concentration in the LFG (F) and the molecular weight ratio of CH_4 and carbon (16/12).

$$DDOC_m = W \bullet DOC \bullet DOC \bullet MCF \quad (2.4)$$

$$L_o = DDOC_m \bullet F \bullet 16/12 \quad (2.5)$$

The study requires CH_4 estimations for several years since opening and after closure of the landfill sites. Therefore time needs to be a consideration in the calculations. The following equations are used in the IPCC model to estimate emissions. MCF accounts for the fact that unmanaged landfills produce less CH_4 from a given amount of waste than anaerobic managed landfills because in unmanaged landfills a larger fraction of waste decomposes aerobically in the top layer.

$$DDOC_{ma_T} = DDOC_{md_T} + (DDOC_{ma_{T-1}} \bullet e^{-k}) \quad (2.6)$$

$$DDOC_{m\ decomp_T} = DDOC_{ma_{T-1}} \bullet (1 - e^{-k}) \quad (2.7)$$

$$CH_4\ generated_T = DDOC_{m\ decomp_T} \bullet F \bullet 16/12 \quad (2.8)$$

Where T represents the inventory year $DDOC_{ma_T}$ the accumulated $DDOC_m$ at the end of year T , $DDOC_{ma_{T-1}}$ is the amount of $DDOC_m$ accumulated at the end of year $T-1$, $DDOC_{md_T}$ is the $DDOC_m$ deposited in the landfill in year T and $DDOC_{m\ decomp_T}$ is the $DDOC_m$ decomposed in year T . The amount of CH_4 formed ($CH_4\ generated_T$)

is calculated by multiplying the CH₄ fraction (F) and the CH₄ /C molecular weight ratio. Since LFG usually consists of approximately 50% CH₄ and 50% CO₂ the IPCC default value for F is 0.5. Only waste that contains large amounts of fat or oil can generate LFG with higher CH₄ concentrations.

The IPCC Waste Model 2006 is based on Equations 2.7 and 2.8. To calculate the amount of DOC decomposing to CH₄ and CO₂ each year, the model keeps a running total of the amount of DDOC_m in the disposal site, taking account of the amount deposited each year and the amount remaining from previous years. A time delay between deposition of the waste and the start of CH₄ generation can also be defined. The amount of CH₄ generated from the DDOC_m is subtracted from the CH₄ recovered and CH₄ oxidised in the cover material to give the amount of CH₄ emitted to the surface of the landfill (IPCC, 2006).

The model can also be run if composition data are not available. It then uses the single-phase model based on bulk waste where DOC can be a weighted average of the waste in the landfill. The following equation estimates DOC using default carbon content values:

$$DOC = \sum_i (DOC_i \bullet W_i) \quad (2.9)$$

Where DOC is the fraction of organic carbon in bulk waste, DOC_i is the fraction of organic carbon in the waste type *i* (for example the wet weight default value for paper is 0.4) and W_i is the fraction of waste type *i* by waste category (IPCC, 2006). Depending on the data that is available both options can be used and when waste

composition is relatively stable, both options give similar results. However, when rapid changes in waste composition occur, options might give different outputs. Both options can be used for estimating the long-term storage of carbon in harvested wood products (HWP) in landfills (IPCC, 2006).

CH₄ emissions are calculated (equation 2.10) by subtracting the CH₄ gas recovered from the landfill and the CH₄ oxidised in cover soils.

$$CH_4 \text{ emitted}_T = \left(\sum_x CH_4 \text{ generated}_{x,T} - R_T \right) \bullet (1 - OX_T) \quad (2.10)$$

R_T represents the CH₄ recovered in year T. OX represents the amount of CH₄ from landfills that is oxidised in cover soils. The default value for OX is zero, however, 0.1 is justified for covered, well managed landfills. Any value higher than 10% must be justified (IPCC, 2006).

California Landfill Methane Inventory Model (CALMIM)

FOD models have been used for several years to estimate LFG production at landfill sites. Initially they were used to predict and quantify the potential for LFG production from a landfill to assess the necessity for a landfill gas extraction system. Most models however do not include cover soils characteristics, seasonal variability of CH₄ oxidation rates in cover soils and the effect of gas extraction systems on CH₄ profiles in cover soils (Bogner *et al.*, 2014). The California Landfill Methane Inventory Model (CALMIM) takes all of the above into consideration when modelling CH₄ emissions from landfill sites.

It was originally field validated for California between 2007 and 2010 and then again in 2011-2013 which included international sites to significantly improve the models applicability globally. CALMIM is different from FOD models in that it is a theoretical prediction of CH₄ emissions based on 1-Dimensional model representing CH₄ transport and oxidation and calculates annual site-specific CH₄ emissions taking into account the various processes that have a controlling effect on emissions from landfill sites.

These include the surface area and properties of the cover materials, the percentage of gas extraction across the landfill site and the climatic factors that affect seasonal CH₄ oxidation in cover soils. CALMIM is freely downloadable at <http://www.ars.usda.gov/services/software/download.htm?softwareid=300>.

CALMIM conforms to Tier 3 of the IPCC methodology for measuring CH₄ emissions as it is based on measured country specific parameters.

Based on the site specific information submitted by the user CALMIM will calculate annual emissions for each cover type and then provide the sum as a total of annual CH₄ emissions. Due to the fact that both transport and oxidation of CH₄ is linked to climatic conditions global models TEMPSIM (air temperature simulation), RAINSIM (precipitation simulation), SOLARCALC (solar radiation simulation) and STM² (soil temperature and moisture model) are linked to CALMIM. The user can input site-specific details on cover types, that is, daily, intermediate and final covers. A number of materials used for cover can also be chosen from the model. Cover thickness varies from 2.5 cm to 2.5 m. Default values can be used if site specific information is not available.

The model calculates CH₄ concentration using Equation 2.11 as:

$$\text{CH}_4 \text{ Base} = (\text{CH}_{4_Default}) (1 - 0.3 * \text{Coverage } \%) \quad (2.11)$$

CH₄ Base is the CH₄ concentration in the specific cover type chosen by the user, CH₄_Default is the default cover concentration based on the type of cover material input into the model and Coverage % is the percentage gas extraction currently in existence within that cover type (ranging from 0 – 100%). If the default value is not used this equation is not applicable as the user input value will just be applied by the model. The 30% reduction in emissions if a gas extraction system covers the surface 100% is a conservative value based on field observations (Spokas *et al*, 2011). Just like other models CH₄ emissions can be calculated with and without oxidation. Based on the cover type chosen CH₄ oxidation rate varies from 1 µg CH₄ g_{soil}⁻¹ d⁻¹ (for daily cover) to 200 µg CH₄ g_{soil}⁻¹ d⁻¹ (for final cover). Based on over 900 laboratory studies conducted for each cover type the CH₄ oxidation rate is divided into nodes and time specific oxidation rates and is a function of soil temperature and soil moisture in each node (Figures 2.2 and 2.3) (Spokas *et al*, 2011).

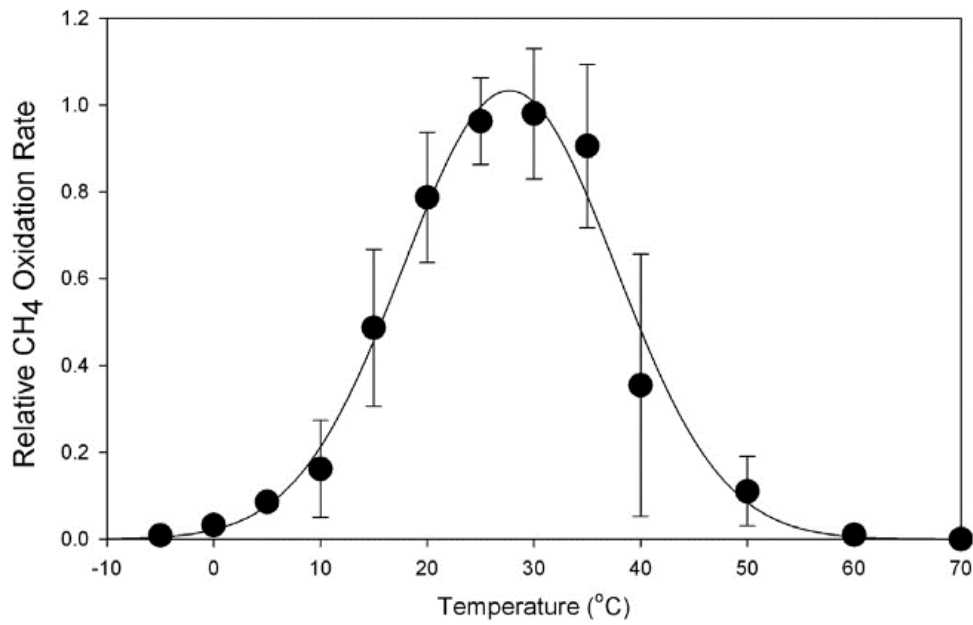


Figure 2.2: Effects of soil temperature on CH_4 oxidation (Spokas and Bogner, 2011).

The California sites used in the initial CALMIM validation showed oxidation rates of $112 \mu\text{g CH}_4 \text{ g}_{\text{soil}}^{-1} \text{ d}^{-1}$ – $644 \mu\text{g CH}_4 \text{ g}_{\text{soil}}^{-1} \text{ d}^{-1}$ with an optimal temperature of 27.6°C and soil moisture of -33 kPa (Spokas *et al*, 2011, Spokas and Bogner, 2011).

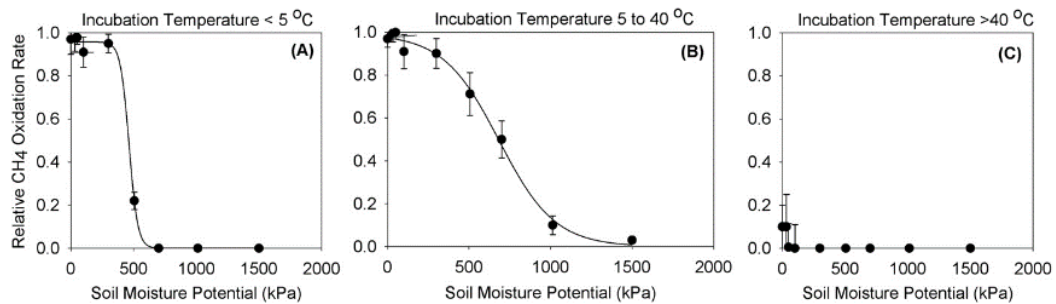


Figure 2.3: Effect of soil moisture content on relative rates of CH_4 oxidation as a function of soil temperature (A) $< 5^\circ\text{C}$, (B) $5\text{--}40^\circ\text{C}$, and (C) $> 40^\circ\text{C}$ (Spokas and Bogner, 2011).

Model Outputs	Model outputs are written directly to Excel compatible files for each cover type		
	Daily Surface CH ₄ emissions	With oxidation Without oxidation	g CH ₄ m ⁻² d ⁻¹ g CH ₄ m ⁻² d ⁻¹
Soil Nodes (2.5 cm layer in cover)	Soil Temperature		°C
	Soil Moisture		Volumetric (cm ³ cm ⁻³)
	Air-filled porosity		cm ³ cm ⁻³
	Oxygen Concentration		% O ₂
	CH ₄ Concentration	With oxidation Without oxidation	% CH ₄ % CH ₄
	CH ₄ oxidation rate		g CH ₄ m ⁻² d ⁻¹
	CH ₄ oxidation percentage		%
	Bulk density		g cm ⁻³
	Fraction of time oxidizing		0 to 100% (0-1)
	Simulated Weather Data		
	Maximum air temperature		°C
	Minimum air temperature		°C
	Precipitation		mm

Figure 2.4: Model output parameters provided by CALMIM. Output can be exported as an excel spreadsheet. (Spokas et al, 2011; Bogner et al, 2014).

Bogner et al, (2014) also used CALMIM to simulate CH₄ emissions from landfill sites under future climate change scenarios using the IPCC Special Report on Emissions Scenarios (SRES: scenarios A2 and B1 for 2020, 2050, and 2100). All scenarios showed a decrease in emissions associated with an increase in thickness of cover used at the sites. However, the modelled emissions from Cairo showed an increase for all cover thicknesses mainly due to an increase in temperatures and decrease in moisture at the sites. This highlights the importance and effectiveness of cover materials in decreasing CH₄ emissions from landfill sites. In fact, the study showed that certain areas at the site with only intermediate cover soils accounted for 95% of CH₄ emissions. Therefore, ensuring that waste is adequately covered for all climatic scenarios should decrease the potential for future CH₄ emissions (Bogner et al, 2014).

Input data used for modelling LFG

A number of parameters were required as input into the models. It is recommended that site specific data be used but when this data is not available model default values can be used. However, default values should be used with caution as these values are set according to different practices and standards of landfilling of waste as compared to South Africa.

The following data is needed:

- Landfill latitudinal/longitudinal location
- The year operational landfilling began and waste density.
- The landfill capacity or available airspace at the landfill sites. Airspace is calculated from the total tonnage of waste that can be accepted at the landfill and includes the volume of cover material that will be used throughout the life of the landfill.
- The annual tonnages of waste disposed at the site and its composition including the fraction of domestic waste deposited.
- Values for k , L_o , the CH_4 and CO_2 fractions and the CH_4 oxidation factor.
- Cap and liner details at the landfill sites including thickness, porosity and infiltration rates.
- Hydraulic conductivity of liquid in the landfill, the absorptive capacity and leachate recirculation.
- Cover material and thickness for daily, intermediate and final covers

- Percentage of gas extraction coverage at the landfill site

Total annual tonnages of waste disposed at the landfill sites were obtained from the site operators as recorded by weighbridges at the site. Waste data for City of Joburg landfill sites were only available from 2001. The total waste (available data and projected data) was compared to the available remaining airspace for all landfills to ensure accurate representation of total waste placed.

The IPCC Waste Model requires waste data for more than 50 years by default with the start year of filling set at 1950. For this study, historical data from 1950 is not available as the oldest landfill used in this study opened in 1988. Hence, for individual sites the years prior to opening were zeroed. However, for national LFG estimates historical data can be estimated proportionally to the growth in population, economic indicators or a combination of both. Changes in waste management policies to reduce waste generation and to promote alternatives to landfill disposal should be taken into account in the analyses. In South Africa, only a few landfill sites were operational since 1950 with very minimal to non-existent data being available on the operation of the sites. Hence, this study will focus on emission contribution from landfill sites in operation since 1980 using population statistics and waste collections trends to determine emissions from waste disposal.

Previous studies have found that the waste collected from urban areas in South Africa is highly variable in composition. Waste collected from relatively affluent suburbs is typical of that of most developed countries (Fourie and Morris, 2004). A study by Fourie *et al.*, (1997) characterised waste streams from a relatively affluent

suburb (Benoni) and an informal settlement (Wattville). The waste from Wattville was found to contain a high percentage of ash (due to the residents' use of coal for heating and cooking). The study divided waste streams in South Africa into two categories namely 'rich' waste from affluent areas and 'poor' waste from non-affluent areas. Builders rubble is usually used as daily cover material. Most sites have a composting area for garden waste and only a fraction is landfilled.

Landfills used in this study accept waste from both affluent and non-affluent areas. Therefore, a combination of the waste compositions and fractions defined by IPCC Waste Model as well as waste characterisation studies completed for the City of Johannesburg is used in this study (Figure 2.5).

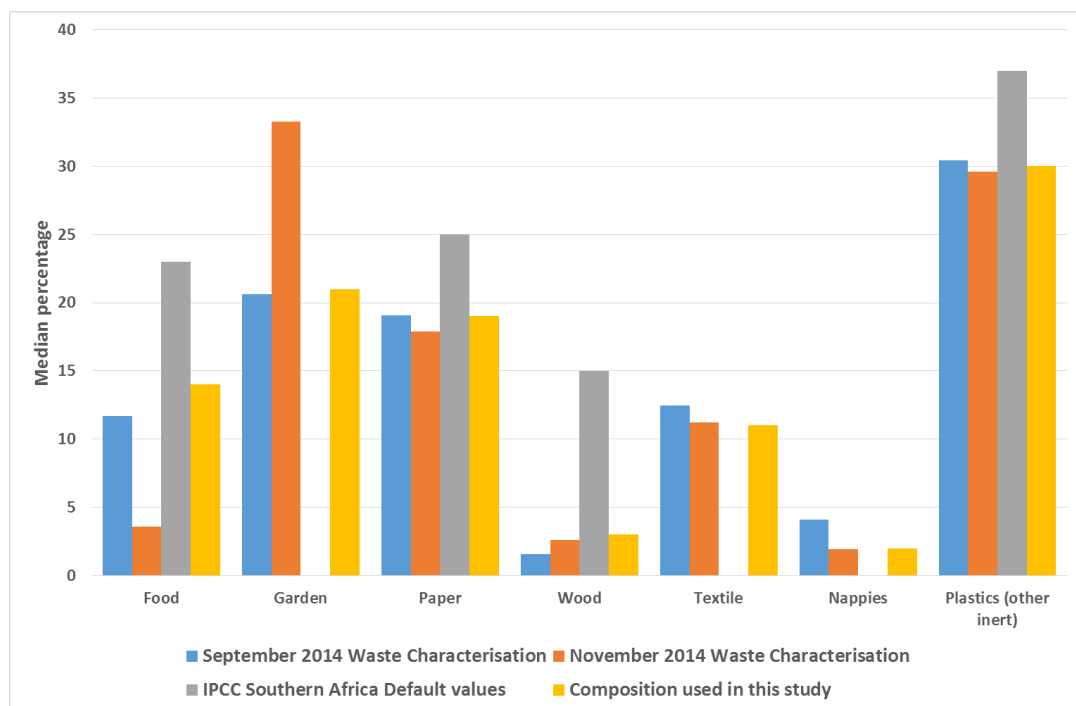


Figure 2.5: Waste composition percentages accepted at the landfills.

The IPCC default compositions for Southern Africa, as supplied by the model, is also included for comparison. Cap and liner details for the landfills were obtained

from the Minimum Requirement documents for waste disposal in South Africa (Figure 2.6). All permitted landfills are required to follow the guidelines set out in the Minimum Requirements. Unlined cells at specific sites were considered and these were included as containing no liner in the model.

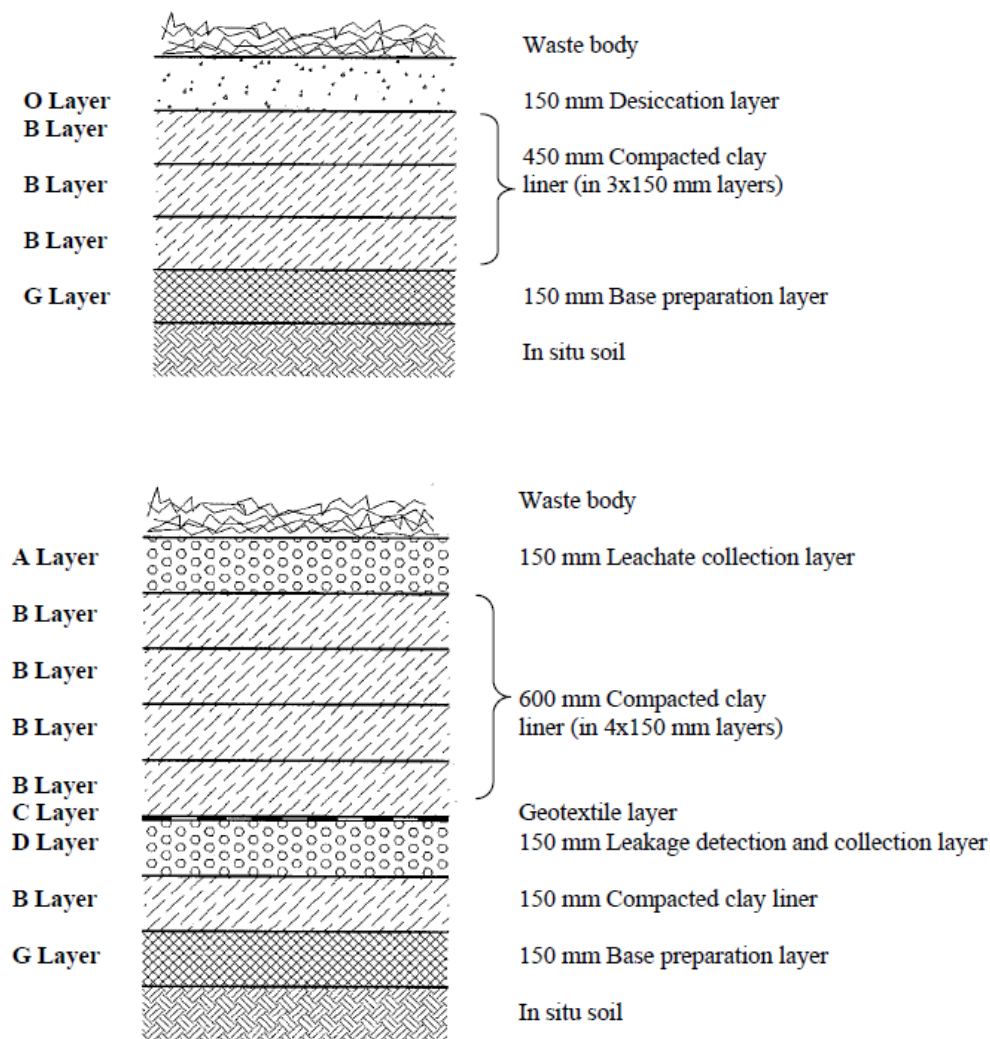


Figure 2.6: Liners used at landfills in South Africa. A triple clay layer is used for GLB⁻ landfills (top) and composite layers including a geotextile layer is used in a GLB⁺ landfill site (DWAF, 1998).

Both default and site specific values were used for k , L_o and the CH_4 and CO_2 fractions in all models. Values for k were obtained from the IPCC Guidelines for Greenhouse Gas Inventories (IPCC, 2006). $k = 0.07$ is used for rapidly degradable waste (food and sewage sludge) and $k = 0.04$ for slowly degradable waste (paper and textiles). CH_4 and CO_2 fractions were assumed to be 55% and 45% respectively based on monitored gas generation data at the sites. The IPCC value of 10% (0.1) was used to represent the CH_4 oxidation factor.

Hydraulic conductivity, absorptive capacity and leachate recirculation were also obtained from the Minimum Requirements (Table 2.2). These factors are important considerations as the flow rate of water and leachate through the liner is dependent on the hydraulic conductivity and absorptive capacity of the soil. (Pitchel, 2005). GaSim and CALMIM allows for the inclusion of gas recovery at the sites. The models were run without gas recovery to allow for the comparison with LandGEM and IPCC that does not allow for the inclusion of recovery systems and to compare to actual gas generation at the sites from the gas extraction systems installed.

Table 2.2: Model input data used for this study. Where possible site specific data is used. Default values are highlighted in yellow.

Parameters		Class B (GLB ⁻)	Class B (GLB ⁺)
Infiltration (mm/yr)		60 - 120	60 – 120
Cap Type		Soil, builders rubble and ash	Soil, builders rubble and ash
Daily		0.15 m	0.15 m
Intermediate		0.3 m	0.3 m
Final		0.5 m	0.5 m
Liner Type		composite (1.2 m)	triple clay (0.75 m)
Waste density (t/m³)		0.75 – 1.1	0.75 – 1.1
Effective Porosity (%)		1 – 20	1 – 20
Hydraulic conductivity (m/s)		1 x 10 ⁻⁹	1 x 10 ⁻⁹
Absorptive Capacity (% v/v)		1 – 5	1 – 5
Gas fractions (%)	CH₄	55	55
	CO₂	45	45

Landfill	% gas recovery	% vegetation cover	% organic material in cover
Weltevreden	24	10	5
Rooikraal	35	20	5
Simmer and Jack	68	40	5
Rietfontein	27	5	5
Robinson Deep	35	5	5
Marie Louise	25	20	5
Ennerdale	0	5	5
Linbro Park	0	70	5
Goudkoppies	0	20	5

Subsurface gas generation monitoring

In order to obtain gas generation at the landfill sites a number of subsurface probes were placed on the boundary of the landfill sites, as part of the thesis. A survey conducted in September 2008 highlighted the need for the design of the probes to be changed as most probes at the sites were blocked, damaged or scavenged. The new probes were developed and installed in 2009. They are perforated allowing for gas movement and range in depth from 1-3m along the perimeter of the site. They contain no metal and therefore are not at risk of being stolen for scrap metal (Figure 2.7).

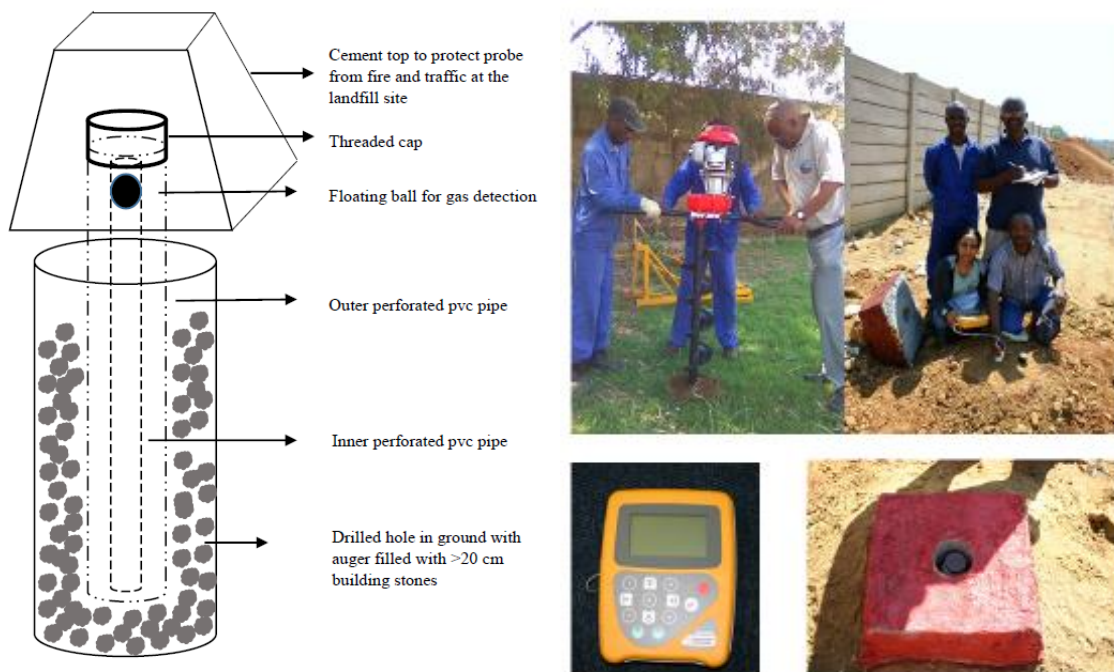


Figure 2.7: Design of landfill gas probe installed at all sites (left). Right shows the drilling of holes using an auger to insert probes and monitoring of gas after probe installation using a hand-held LFG analyser.

Direct readings of gas concentrations (CH_4 , CO_2 , O_2 , H_2S and CO) are taken using the GA2000 hand-held infrared gas analyser developed by Geotec. The hand held

analyser measures CH₄ and CO₂ by dual wavelength infrared cells and O₂ by internal electrochemical cell. Sampling was conducted on a quarterly basis (once every three months) in accordance with the Minimum Requirements (DWAF, 1998). Concentrations were logged onto a built-in data logger on the analyser. Twenty-seven probes were installed at Goudkoppies (Figure 2.8), twenty-five at Marie Louise (Figure 2.9, top), thirty-one at Robinson Deep (Figure 2.9, bottom), forty seven at Linbro Park (Figure 2.10, top) and twenty seven at Ennerdale (Figure 2.10, bottom). Distances between probes were determined to ensure the entire site is covered.



Figure 2.8: Location of subsurface gas probes installed along the boundary of Goudkoppies landfill site (n=27).

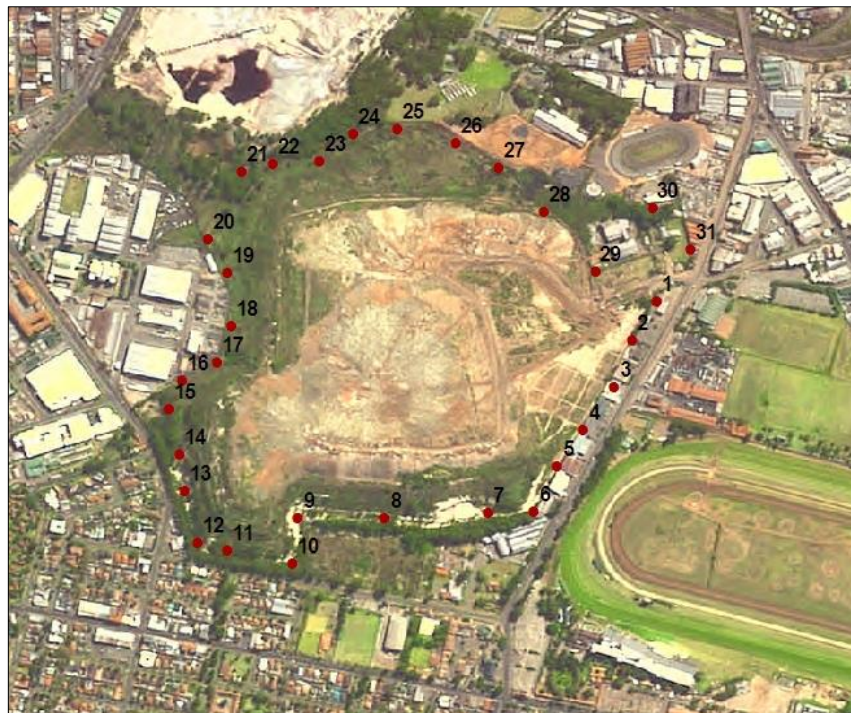


Figure 2.9: Location of subsurface gas probes installed along the boundary of Marie Louise (top), $n = 25$ and Robinson Deep (bottom), $n = 31$.



Figure 2.10: Location of subsurface gas probes installed along the boundary of Linbro Park (top), $n = 47$ and Ennerdale (bottom), $n = 27$.

Flux chamber measurements

To meet the objectives of the study a number of methods to measure surface emissions of a landfill surface were considered. Emissions can be calculated from concentration profiles in soil cores; spatially distributed static and dynamic closed chambers (Bogner and Scott, 1995; Maurice and Lagerkvist, 1997; Cossu et al., 1997); eddy-covariance or micro-meteorological methods can give concentration profiles on top of the landfill from which whole landfill emissions can be determined (Oonk and Boom, 2000; Savanne et al., 1997); and measurements at a distance from the landfill (using a tracer gas to detect gas plumes) can be used to obtain emissions from the entire landfill (Czepiel et al., 1996). No one method is standardised in the literature and each method has its associated advantages and disadvantages.

The application of three methods (flux chambers, Open Path Fourier Transform Infrared Spectroscopy (OP-FTIR) and a micro-meteorological method) to measure surface LFG emissions at a selected landfill was originally proposed. However, the application of an FTIR became too cumbersome and expensive to implement hence LFG emissions were monitored using static flux chambers and a mobile caravan, equipped with ambient air quality gas analysers. This allowed for LFG to be monitored both spatially across the landfill surface and temporally.

Flux chambers to measure CH₄ and CO₂ fluxes measured at the surface of Robinson Deep and Marie Louise landfill sites were placed as presented in Figure 2.11. A transect at the Robinson Deep landfill site was chosen and diurnal measurements

were taken at specific points thorough the day (Chambers 1 to 7), from 6:00 to 20:00 over 3 days, to obtain a diurnal pattern of emissions from the surface. This method was chosen since the chamber covered a very small area on the surface of the landfill which meant a large number of points was needed to obtain whole site emissions. Samples were taken with the passing of a cold front to assess the impact of decreasing pressure on LFG emissions. Random sampling points (conducted on the 27 July 2011 between 10:00 and 16:00) across the landfill site were also taken including areas where gas well installations had taken place (Chambers 1 to 15). Random sampling at Marie Louise landfill site was also conducted (Chamber 1 to 8, Figure 2.11). However, the data set is incomplete due to an armed robbery that took place at the site during monitoring, after which it was considered unsafe to continue sampling at the site. The data that were obtained will be presented.

The chamber shown in Figure 2.12 (a) is constructed from stainless steel. It has a 10 cm sharpened base which allowed for it to be knocked into the hard soil at the landfill site and ensuring that the sample area was fully concealed. Chambers made from PVC piping were originally used and were unable to seal due to the hard surface at the sites hence a new chamber made from stainless steel was designed and constructed specifically for this project. The LFG samples were taken using glass syringes with a lock (Figure 2.12 (b)) at 5, 10 and 15 minute intervals. Air pressure, altitude and temperature were also measured at each sampling point using a hand held kestrel pocket weather tracker (Figure 2.12 (c)). Gas emission fluxes (φ) were calculated at each point in $\text{g/m}^2/\text{day}$ using Equation 3.1 as follows:

$$\varphi = \left(\frac{V}{A}\right) \left(\frac{dC}{dt}\right) \quad (2.12)$$



Figure 2.11 Location of flux chambers placed at the surface of the Robinson Deep (top) and Marie Louise (bottom) landfill sites.

where V is the enclosed chamber volume, A is the enclosed area, dC/dt is the rate of increase in gas concentration in the chamber with time. The value of the square of the correlation coefficient, R^2 , for the dC/dt term used in the above equation is an important indicator of data quality and, for this reason, R^2 is used as an acceptance criterion for emissions measurements. Previous studies have used an $R^2 > 0.5$ in the early 1990's (Bogner et al, 1993) and later $R^2 > 0.8$ (Bogner et al, 1997). Fourie and Morris (2004) disregarded values with $R^2 < 0.5$ for previous flux measurements at landfills in South Africa. This study disregarded all measurements with $R^2 < 0.9$ at Robinson Deep and $R^2 < 0.8$ at Marie Louise.

Both CH_4 and CO_2 emissions were measured using the flux chambers. Measuring both CH_4 and CO_2 provides a data set that gives insight into CH_4 generation and oxidation rates. The SRI Gas Chromatography (GC) unit was used to analyse the samples taken (Figure 2.13 (a)). The GC was equipped with a Restek Packed, Propak 6feet, 2 mm stainless steel column (Figure 2.13 (b)). An inlet was used to inject a 2 ml sample into the GC at a time. Helium gas was used as a carrier with a flow rate of 15 ml/min through the column. The GC was modified to include a methaniser which allowed for the measurement of CO_2 (Figure 2.13 (c and d)). The methaniser enables the Flame Ionization Detector (FID) to detect low levels of CO_2 . It is packed with a nickel catalyst powder and is heated to $380^\circ C$ with the FID. Sample temperature was set at $50^\circ C$. When a sample is injected into the GC it passes through the FID and then through the methaniser which converts CO_2 to methane. Since the conversion of CO_2 to methane occurs after the sample has passed through the column, their retention times are unchanged. Hence during analysis, methane would be represented by the first peak and CO_2 by the second.

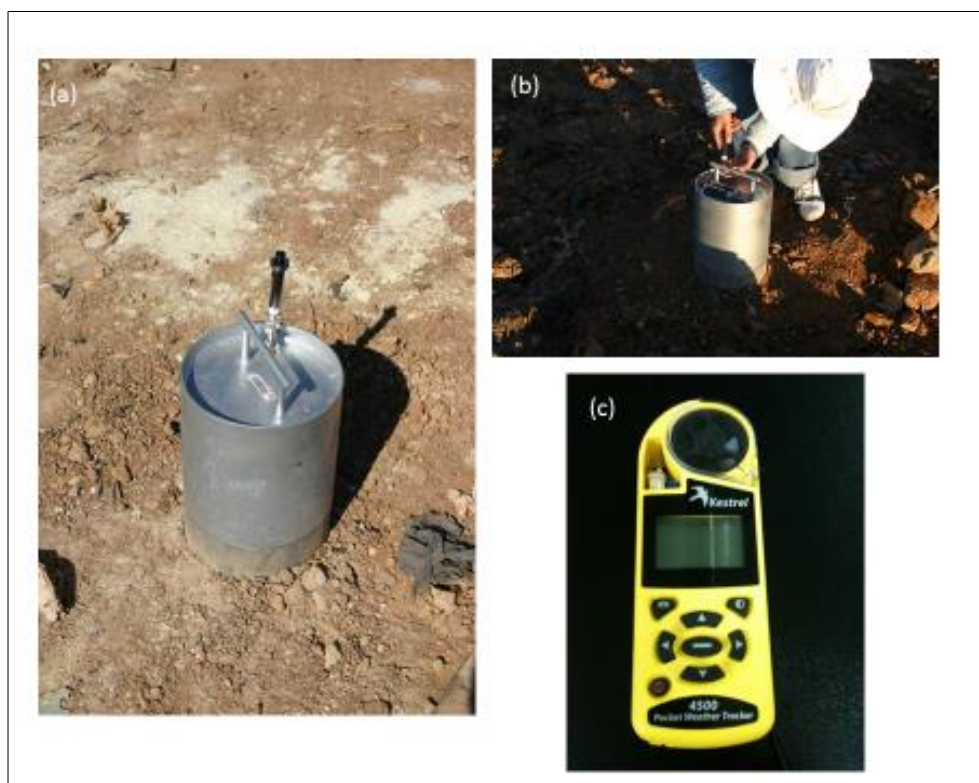


Figure 2.12: (a) flux chamber used to measure CH_4 and CO_2 emissions, (b) sample taken during monitoring campaign and (c) kestrel used to measure ambient conditions at each sample

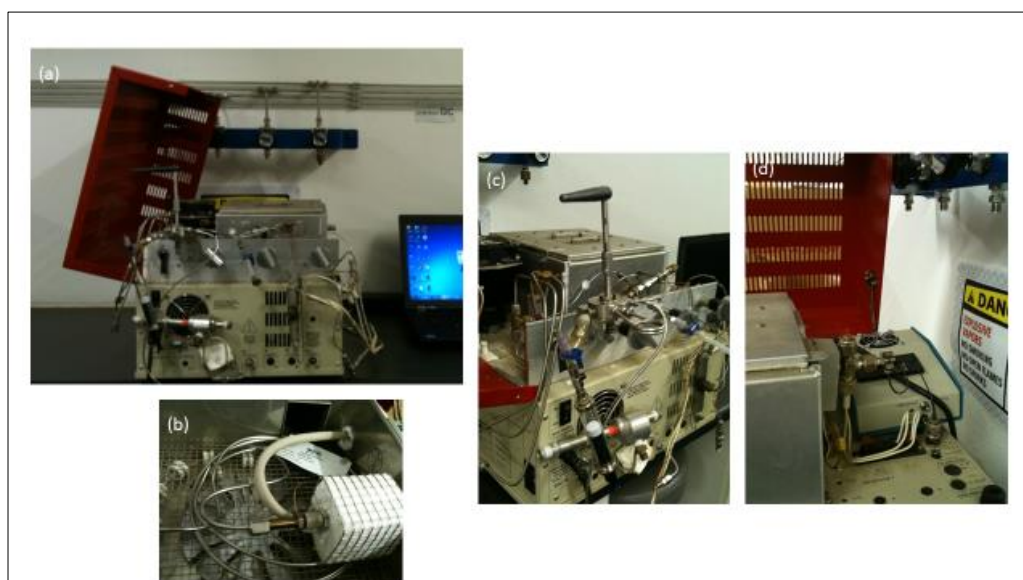


Figure 2.13: SRI GC used to analyse samples collected from flux chambers, (b) packed column used for analysis (c) specially designed inlet to inject 2ml of sample (d) methaniser to enable CO_2 to be measured in the sample.

The spatial distributions of the CH₄ and CO₂ from the landfills are presented in contour plots. It is important to recognize that the gas concentrations are only correct at the probe. Other areas are interpolated values calculated using a grid feature on the program, Surfer, a grid-based contour program. Gridding is the process of using original data points (observations) in an XYZ data file to generate calculated data points on a regularly spaced grid (in this case, X and Y represents the latitudinal and longitudinal locations of the probes and Z the concentration of the gases). Interpolation schemes estimate the value of the surface at locations where no original data exist, based on the monitored data values. Surfer grids the input data and generates the contour map. The kriging gridding method on Surfer was used for creating the contour plots to represent CH₄ and CO₂ fluxes at the landfill surface.

Ambient air quality monitoring and source apportionment

Ambient air quality monitoring was undertaken using a mobile caravan for the period May to August 2010 and February to June 2011 at the Robinson Deep landfill site. Direct surface measurements were conducted in July 2011 and do not coincide with the air quality monitoring period. Pollutants measured include SO₂, NO and NO₂, O₃, CO, PM_{2.5}, H₂S, BTEX (Benzene, toluene, ethyl-benzene and Xylene). CH₄ and Non-Methane Hydrocarbons (NMHC) were also monitored for this period. Meteorological parameters were recorded including wind speed, wind direction, temperature, solar radiation and atmospheric pressure. The monitored concentrations of pollutants were analysed and where possible compared to the South African ambient air quality standards to

determine compliance and the potential for impact on the environment. To ensure the safety of the monitoring station, it was placed at City Power cable yard and the Turffontein racecourse. These sites were also chosen due to odour and health complaints received from the employees working on the premises. The City Power offices are directly adjacent (50 meters from the landfill boundary) on the western border of the landfill site and the caravan was placed approximately 160 meters from the landfill boundary within the Turffontein Racecourse (Figure 2.14).

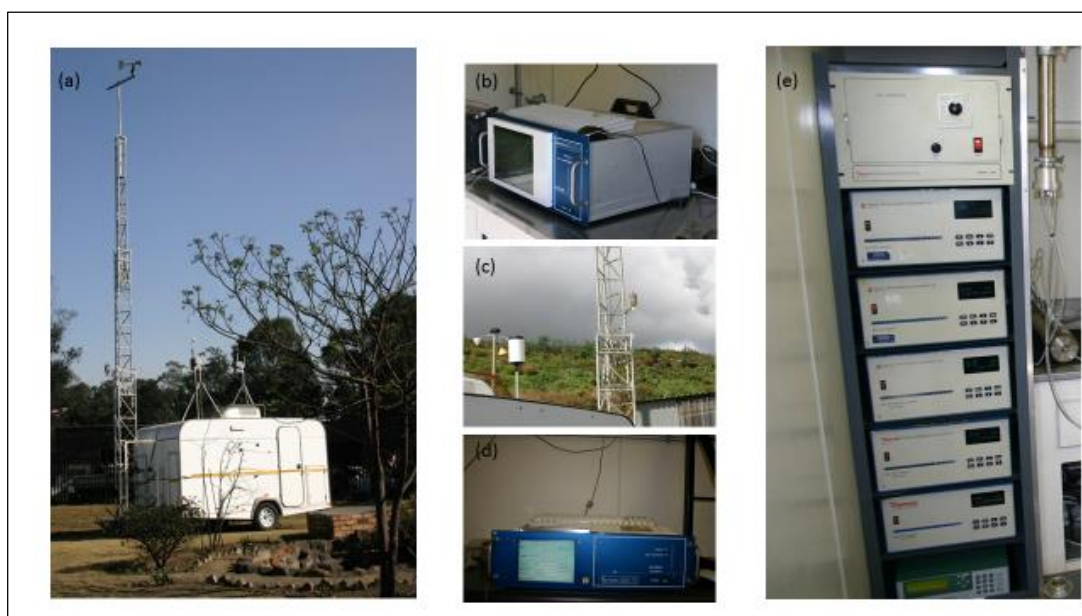


Figure 2.14: (a) Mobile caravan set up used at City power and Turffontein Racecourse, (b) Synspec BTEX GC 955 placed in caravan for continuous measurements, (c) inlet, rain gauge and weather mast outside mobile station with Robinson Deep landfill site in the background, (d) Synspec CH₄ and NMHC Alpha 115 analyser and (e) air pollution analysers placed in mobile station for continuous monitoring.

The openair package within the R software program was used in this project to analyse the air pollution and meteorological data collected during the ambient air pollution monitoring in winter and summer. Developed as part of a Natural

Environment Research Council (NERC) project, openair is aimed at providing open source software for the analysis of air pollution data. The software is can be downloaded at <http://cran.r-project.org/web/packages/openair/index.html>.

The software is based on the Conditional Probability Function (CPF) which calculates the probability that the concentration of a particular pollutant is greater than a specified value in a given wind sector (Uria-Tellaetxe and Carslaw, 2014). However, it adds to this method by including the dependence on wind speed for pollution dispersion using the bivariate polar plot thus providing a new method termed the Conditional Bivariate Probability Function (CBPF). This allows sources to be determined based on their wind speed, for example, buoyant plume emissions from stacks tend to peak under higher wind speed conditions whilst non-buoyant plumes such landfill sites, tend to have highest concentrations under low wind speed conditions (www.openair). Thus openair was used to determine the impact of the landfill site on the surrounding environment using the monitored data from the mobile caravan and comparing it to modelled source emissions.

Results from openair are compared to dispersion modelled emissions using the Atmospheric Dispersion Modelling System (ADMS). ADMS consists of a suite of models and was developed by the Cambridge Environmental Research Consultant (CERC) and has been validated against numerous global field data sets. ADMS 4 was used for the air quality assessment in this study. A maximum of 300 sources can be entered into the model where 30 line sources, 30 area sources and 30 volume sources can be modelled simultaneously.

Dispersion models require input data including meteorological data (for example wind speed and direction, temperature, humidity) and emissions data such as source location and height, diameter and exit velocity, temperature and flow rate. Terrain also needs to be considered especially areas of complex terrain which would affect wind speed and turbulence, key components for dispersion. Meteorology is an essential requirement and is the principal factor for the dispersion of pollutants in the atmosphere. Important meteorological factors that directly impact the dispersion of a pollutant include vertical profiles of wind speed and direction, atmospheric turbulence and ambient air temperature. Hourly prognostic meteorological data (including wind speed and direction; temperature, humidity, precipitation and cloud cover) from the South African Weather Service (SAWS) were used for the modelling, measured at OR Tambo International airport. The landfill sites were modelled as an area source. Predicted pollutant and dust fall out concentrations are compared to the air quality guidelines and the possible impacts within associated buffer zones assessed. Buffer zones around the site at 200m, 500m and 1000m were included in the analyses to justify the requirement and placement of a buffer zone around the landfill site (Figure 2.15).

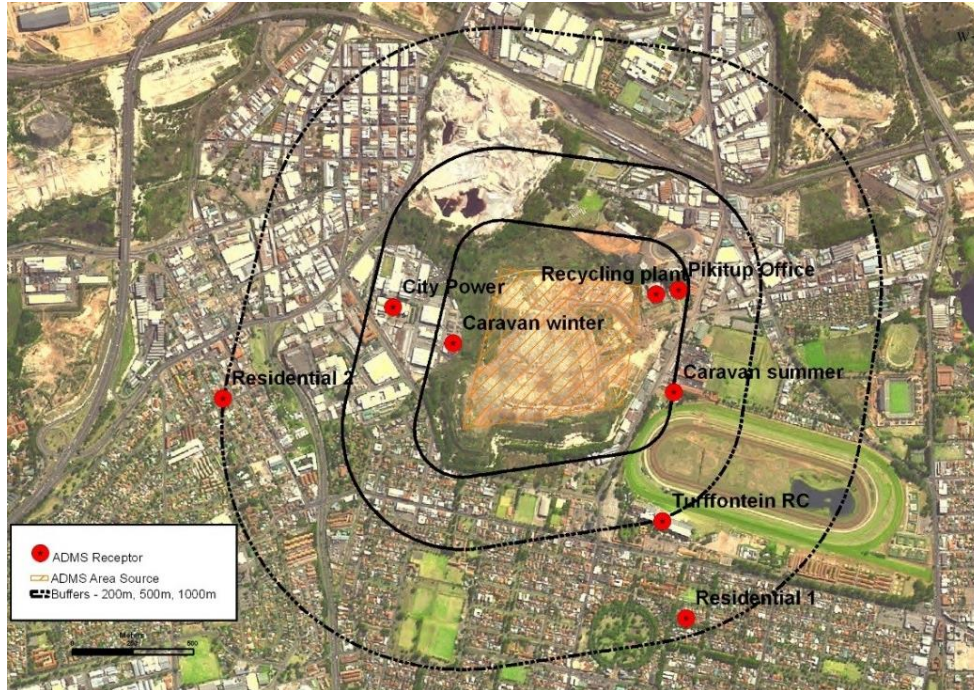


Figure 2.15: Robinson Deep landfill site. The landfill buffer zones are highlighted along with the ADMS receptor points and area source (landfill site).

Emissions factors were calculated for all pollutants at the site. The United States Environmental Protection Agency's (USEPA) LandGEM model was used to determine emission rates at the landfill due to the model having the highest emissions thus representing the “worst” case scenario for emissions. The AP 42 emission factor equation for wind erosion from stockpiles (applicable to landfill sites) and , the PM10 (Particulate Matter, $\mu\text{g}/\text{m}^3$) and TSP (Total Suspended Particles, $\mu\text{g}/\text{m}^3$) were modelled using Equation 2.13:

$$E_{TSP} = 1.9 \times \left(\frac{s}{1.5} \right) \times \left(\frac{365-p}{235} \right) \times \left(\frac{f}{15} \right) \quad (2.13)$$

Where s is the silt content (%), p the number of days when rainfall is greater than (0.25 mm) and f the percentage of time that wind speed is greater than 5.4 m/s. Dispersion simulations of daily average pollutant concentrations were

undertaken for 2011 and 2012. Dust fall out was simulated by using the dry deposition module in ADMS 4. Dry deposition occurs when pollutants are transported from the plume to the surface of the ground with a certain deposition velocity. Background concentrations of the pollutants were not included in the modelling and therefore predicted concentrations should be considered as a contribution to background. As a result ambient concentrations will be higher. Ambient air quality monitoring data recorded at the site between March and August 2010 is used to compare to the modelled concentrations.

Landfill gas extraction and utilisation

Currently there are six operational CDM projects in South Africa consisting of ten LFG extraction and utilisation projects. The eThekweni Municipality (Marianhill and Bisasar Road landfill sites), four landfill sites within the Ekurhuleni Metro Municipality (Weltevreden, Rooikraal, Rietfontein and Simmer and Jack landfill sites). Alton landfill site in Empangeni, the Chloorkop landfill site and two landfill sites within the City of Johannesburg (Robinson Deep and Marie Louise landfill sites). The EMM CDM project will be discussed in detail as a case study.

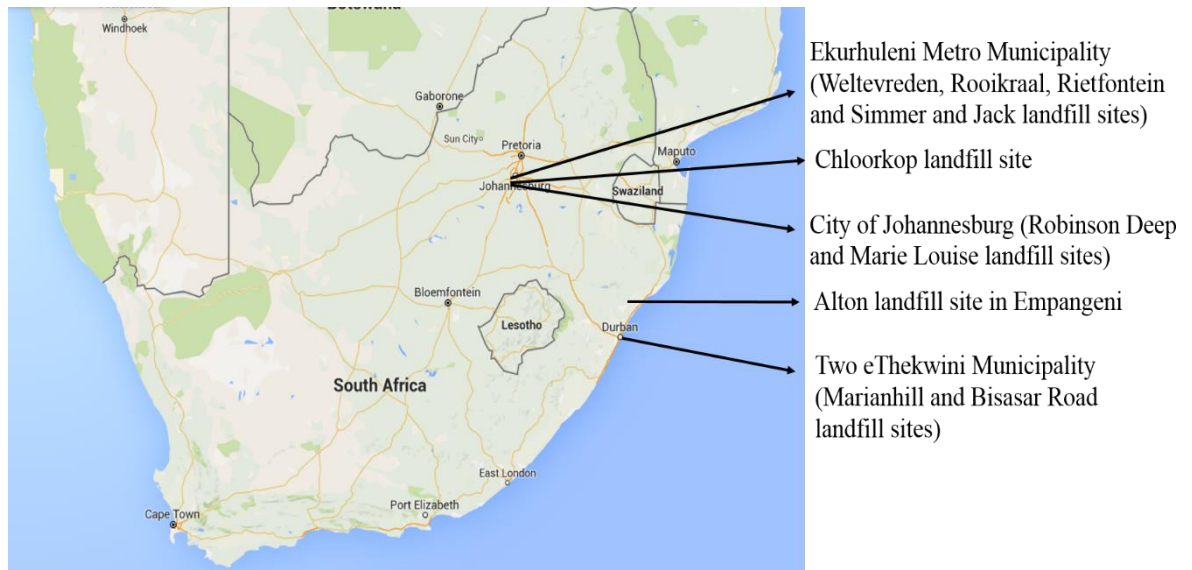


Figure 2.16: Location of registered CDM projects used in this study.

The Ekurhuleni Metropolitan Municipality (EMM) has established a landfill gas extraction and utilisation project at four of its landfill sites. The extraction and utilisation of landfill gas reduces harmful greenhouse gas (GHG) emissions and prevents explosion hazards at the landfill sites from the accumulation of methane gas.

Assessment of the potential for LFG extraction and utilisation at EMM landfill sites began in 2005, with a feasibility study and trial of landfill gas extraction and flaring. The trial consisted of the installation of two to four gas wells and a 300 m³/h flare at each site, which was monitored for a six month period. This provided an initial assessment of gas quantity and quality yields. The study found that gas extraction and power generation were feasible and financially viable under the Clean Development Mechanism (CDM) of the Kyoto Protocol, which would allow for revenue to be obtained from the sale of emission reductions. The decision was made to go forward with the development of a LFG CDM Project. The construction of

the full-scale project started on 29 August 2007. CDM registration was finalised in February 2011 retrospectively to 26 October 2010. The CDM Project was intended to be implemented in 2 phases. In phase 1, the LFG will be extracted and flared. In phase 2, the intention is to use the LFG for power generation and export electricity to the grid. Phase 2 was commissioned in September 2014.

EMM has a significant number of Voluntary Emission Reductions (VERs) and Certified Emission Reductions (CERs) that were generated in (Commitment Period) CP1 but have not yet been issued. The CDM Project's first verification (comprised of all of the VERs generated before registration and all the CERs generated from the time of registration through 29 February 2012) is currently underway. Aenor (Spanish Association for Standardisation and Certification) has been appointed by EMM as the Designated Operational Entity (DOE) to conduct the verification. There are approximately 154 475 VERs and 62 957 CERs that are being verified. Since project inception, emission to the atmosphere of a total of 573 494 tCO₂e (as at 29 February 2015) has been avoided. EMM is currently seeking a buyer for the carbon credits. A number of local and international entities have shown interest in acquiring the credits. Emission reductions generated by the project are expected to be eligible for use as offsets under the proposed Carbon Tax, due to take effect in 2017.

EMM continuously installs horizontal and vertical gas wells as waste is disposed at the landfill sites. Design engineers were appointed for well field design and civil engineering contractor for the gas well installations. The horizontal wells require 10 meters of waste at a minimum before it can be installed. A further 10 meters

need to be filled above the well before the well can be commissioned for gas extraction. Vertical wells are installed once a cell has reached its final landform height. Initially five vertical wells were installed at Simmer and Jack, thirty six vertical wells and two horizontal wells at Rooikraal, eighteen vertical wells and four horizontal well at Weltevreden and eight vertical wells at Rietfontein. Thereafter more wells were installed in five phases over four years. Gas well installations occurred in Phases as follows:

Phase 1: (May 2011 – Jan 2012)

- 17 vertical wells installed at Rietfontein
- 2 horizontal wells installed at Weltevreden
- 1 horizontal well installed at Rooikraal
- 15 vertical wells and 3 horizontal wells installed at Simmer and Jack

Phase 2: (May – August 2012)

- 4 horizontal wells installed at Weltevreden
- 5 horizontal wells installed at Rooikraal
- 1 horizontal well installed at Simmer and Jack

Phase 3: (May – August 2013)

- 6 horizontal wells installed at Weltevreden
- 5 horizontal wells installed at Rooikraal
- 4 horizontal well installed at Simmer and Jack

Phases 4 and 5: (May – August 2014)

- 2 horizontal well installed at Weltevreden

- 2 horizontal wells installed at Rooikraal
- 13 vertical wells at Simmer and Jack

Total gas well installations currently on site are provided in Table 2.3. Additional wells will be installed at all sites when sufficient waste is in place to allow for installations to continue.

Table 2.3: Total gas well installations at EMM landfill sites. Gas well are continuously installed as new cells are developed and/or closed.

	Vertical wells	Horizontal wells	Knock-out pots
Rooikraal			
Phase 1	36	0	7
Phase 2 (A, B)	0	13	3
Weltevreden			
Cell 1, 2, 3	23	4	6
Cell 4, 5	0	13	2
Simmer and Jack			
Cell 1, 2, 3, 4	38	0	7
Cell 5, 6	12	10	1
Rietfontein			
Cell 1,2,3	25	0	2

Gas wells contain a well head that allows for the well to be controlled and monitored. Horizontal wells lead to a manifold box where the well can be opened and closed. Condensate knock-out pots are strategically located at low points in the systems to allow condensate to accumulate and be pumped to the main knock-out at the flare compound (Figure 2.17). The condensate is then pumped to the main leachate pond at the sites. Wells run to the flare via a ring main.



Figure 2.17: (a) Drill rig used to install vertical wells, (b) vertical well installation into drilled hole, (c) and (d) horizontal well installations, (e) knock-out pot, (f) gas well head and (g) manifold box where horizontal well heads are housed.

Flare systems were installed at all sites and were fully operational by December 2008 (Figure 2.18). The flare system includes:

- A gas delivery unit – where the gas to the flare and other parameters are recorded (Figure 2.18 (a))
- A burner – to effectively burn the LFG at $>500^{\circ}\text{C}$ (Figure 2.18 (b))
- A flame arrestor – that controls the lighting of the burners
- A blower – to extract gas from the gas wells at specific pressure (Figure 2.18 (c))
- Actuated louvers – controls air flow to the flares burners (Figure 2.18 (d))
- A condensate knock-out – that allows all condensate collected in the system to drain to a central point (Figure 2.18 (g))

- Flow meters – that monitors and measures the flow of LFG to the flare
- A backup generator – that starts as soon as there is a power failure in the main electricity supply. This allows the flare to operate at all times.



Figure 2.18: (a) Gas Delivery Unit (GDU) within the flare compound, (b) burners, (c) blower, (d) louvers, (e) flare and (f) LFG inlet feeding gas to the flare.

The outlet from the blower passes through a particulate filter to remove any residue still in the gas. The gas flow is recorded by a flow meter as is fed to the gas delivery unit. Two gas lines are present, one line allows gas to be diverted to biofuel engines to produce electricity and another to the flare (for access gas). The burners that flare the LFG are opened upon detection of a pilot flame and is controlled by a slam shut valve. Air flow for combustion is controlled by actuated louvers (Organics Operation and Maintenance Manual, 2007).

The focus at present is on LFG utilisation and installation of gas engines at all sites as new well installations become sufficiently covered with waste to allow gas to be extracted. Furthermore, there is a possibility of using the LFG to generate clean

biogas (methane) fuel to run EMM's car fleet. A pilot project has been implemented since June 2010 to generate and use landfill gas as a vehicle fuel at a closed landfill site in EMM. There is consideration to expand this project to one of the larger operational landfill sites to convert the landfill gas to vehicle fuel. This will be considered in the renewal process of the CDM registration as it would include a change in the monitoring methodology.

The baseline and monitoring methodology approved for the EMM project is the consolidated baseline methodology ACM0001, Version 11: "Consolidated baseline and monitoring methodology for landfill gas project activities".

The methodology ACM0001 refers to the following tools which are also applied to the project:

- "Tool for the demonstration and assessment of additionality" (Version 05.2)
- "Tool to determine project emissions from flaring gases containing methane" (EB 28, Annex 13)
- "Tool to calculate baseline, project and/or leakage emissions from electricity consumption" (Version 01, EB, Annex 7)
- "Tool to calculate project or leakage CO₂ emissions from fossil fuel combustion" (Version 02, EB 41, Annex 11)

- “Tool to determine methane emissions avoided from disposal of waste at a solid waste disposal site”. (Version 04, EB 41, Annex 10)
- “Tool to calculate the emission factor for an electricity system” (Version 2, EB 50, Annex 14)

According to ACM0001 version 11, emission reductions are calculated by subtracting the baseline emissions from the project emissions in a year (Equation 2.14)

$$ER_y = BE_y - PE_y \quad (2.14)$$

ER_y represents the emission reductions in year y in tonnes of CO₂ equivalent, BE_y represents the baseline emissions in year y in tonnes of CO₂ equivalent and PE_y represents the project emissions in year y in tonnes of CO₂ equivalent (tCO₂e/yr)

Baseline emissions are calculated as using Equation 2.15 by subtracting emission reductions that the project produces from reductions that would occur in the absence of the project. It also includes emissions avoided as a result of replacing electricity from fossil fuelled power plants.

$$BE_y = (MD_{project,y} - MDBL,y) \times GWP_{CH_4} + EL_{LFG,y} \times CEF_{elec,BL,y} + ET_{LFG,y} \times CEF_{ther,BL,y} \quad (2.15)$$

BE_y represents baseline emissions in year y in tonnes of CO₂ equivalent (tCO₂e/yr), $MD_{project,y}$ is the amount of methane that would have been destroyed or combusted

during the year in the project scenario (tCH_4), $MD_{BL,y}$ is the amount of methane that would have been destroyed or combusted during the year in the absence of the project due to regulatory and/or contractual requirements (tCH_4), the Global Warming Potential GWP_{CH_4} value for methane for the first commitment period is 21 (tCO_2e/tCH_4), $EL_{LFG,y}$ represents the net quantity of electricity produced using LFG, which in the absence of the project activity would have been produced in power plants connected to the grid or by an on-site or off-site fossil fuel based captive power generation, during year y (MWh), $CEF_{elec,BL,y}$ represents the CO_2 emissions intensity of the baseline source of electricity displaced (tCO_2e/MWh), $ET_{LFG,y}$ is the quantity of thermal energy produced utilizing the landfill gas, which in the absence of the project activity would have been produced from on-site or off-site fossil fuel fired boiler during year y (TJ) and $CEF_{ther,BL,y}$ is the CO_2 emissions intensity of the fuel used by the boiler to generate thermal energy which is displaced by landfill gas based thermal energy generation (tCO_2e/MWh), which is zero for this project.

The calculations for the replacement of electricity emissions by the project is provided in detail in the Project Design Document (PDD), which was approved by the Executive Board of the United Framework Convention on Climate Change (UNFCCC) in 2006. An updated PDD was submitted and approved in October 2014 and can be found at <https://cdm.unfccc.int> under Project 3677: Ekurhuleni Landfill Gas Recovery Project – South Africa. Emission reductions for all sites are calculated using Equation 2.11 using other tools in the methodology. Emissions at all sites are calculated as flaring except for Simmer and Jack where electricity

generation was commissioned in September 2014. Electricity generation will be considered from this date when calculating baseline emissions.

Actual gas yields, is analysed and compared to modelled LFG yields for EMM. Other LFG CDM projects in South Africa are also presented in the study. Other landfill sites are used to compare the success of operating a LFG CDM project in South Africa, based on the gas extraction analysis as well as a cost benefit analysis considering the cost of registration, operating and maintaining the project and verification of emission reductions.

Chapter 3

Landfill gas emissions in South Africa

Fugitive landfill gas (LFG) emissions are determined for South African provinces, the eight major Metro Municipalities and individual landfill sites using LandGEM, GasSim, the IPCC Waste Model 2006 and CALMIM LFG estimation models. LFG production (measured using subsurface probes) at landfill sites in City of Johannesburg and Ekurhuleni are analysed to determine the CH₄ and CO₂ ratio. CH₄ and CO₂ fluxes from the surface of the Robinson Deep and Marie Louise landfill sites are analysed and presented. The global warming potential of the LFG emissions and their contribution to total CH₄ emissions in the South Africa are determined.

LFG emissions estimation in South Africa

The 2012 National Waste Baseline Report shows that South Africa generated approximately 108 million tonnes of waste of which 59 million tonnes was general waste. Generally, the waste generated per capita per day in South Africa is more characteristic of developed countries than a developing country. Generation rates are higher in higher income groups than lower income groups (Arendse and Godfrey, 2010).

According to The Gauteng Waste Minimisation Plan (2009) modelled waste quantities discharged into landfill sites in the province by 2020 will be approximately 6 500 000 tonnes based on current waste quantities and improved waste minimisation at source. However, it is more than likely that this amount will

be exceeded as waste minimisation strategies are implemented slowly and not very effectively. For example, the Polokwane Declaration in 2001 set targets of 50% reduction in waste to landfill by 2012 with a zero waste to landfill by 2022 (DWAF, 2001). These targets have not been met mostly due to the lack of financial resources within local municipalities. The collection of recyclable material is largely an informal activity in South Africa often being led by reclaimers at landfill sites. Hence, most waste generated is disposed at landfill sites. LFG emissions were estimated for each province in South Africa using the IPCC Waste Model 2006. The model is based on population and waste generation rates per capita. Waste generation rates were used from the National Waste Information Baseline Report published by the Department of Environmental Affairs, 2012 (Table 3.1). Waste collection rate per province was used according to the latest General Household Survey (StatsSA, 2013) and population statistics were obtained from StatsSA and projected to 2020 (Table 3.2 and 3.3).

Table 3.1: Waste generated per capita per day for each province in South Africa (DEA, 2012).

Province	kg/capita/day	Waste generated as % of total
Eastern Cape	0.31	4
Free State	0.55	3
Gauteng	2.08	45
KwaZulu-Natal	0.43	9
Limpopo	0.28	3
Mpumalanga	1.42	10
North West	0.19	1
Northern Cape	1.50	3
Western Cape	1.85	20

The percentage of households that receive weekly waste removal in South Africa increased from 56.7% in 2002 to 63.5% in 2013, while the percentage of households

disposing of waste in their own or communal landfill or disposing of their waste anywhere has decreased over the past 10 years (StatsSA, 2013). However, the national statistics do not show the divergences in waste collection between rural and urban areas. Urban areas and metropolitan areas are more likely to receive weekly waste removal services and hence increase the overall percentage nationally. In 2013, 91.9% of rural households relied on their own waste removal and disposal services. The most populated provinces Gauteng and the Western Cape have 91.1% and 93% household waste removal coverage respectively whilst smaller provinces like Mpumalanga (38.2%), Eastern Cape (35.2%) and Limpopo (20.2%) have far lower rates of weekly waste collection. Despite the low rates of waste collection in Mpumalanga it is still the third highest contributor to the total waste generated in South Africa at 10% behind Gauteng (45%) and Western Cape (20%) (Table 3.1).

Waste fractions and composition used to model LFG emissions are given in Chapter 2. Default waste degradation rates were used in the IPCC Waste Model. CH₄ and CO₂ fractions were determined by the monitoring of subsurface gas probes at the landfill sites (Figure 3.1). CH₄ is higher than CO₂, showing that all landfill sites are in the methanogenic phase. The ratio of maximum percentage for winter CH₄:CO₂ is higher than summer with maximum being 1.65:1 in summer and minimum 1.3:1 compared to 1.93:1 and 1.34:1 in winter. An average of 55% methane (CH₄:CO₂, 1.22:1) was used for all modelling of LFG emissions for comparison to previous studies. Summer and winter percentages in both CH₄ and CO₂ with average summer CH₄ at 58.7% and winter 58.4% whilst CO₂ averaged at 40.5% and 38.1% for summer and winter respectively.

Table 3.2: Waste collection percentages per province in South Africa (General Household survey 2013, StatsSA (2014)).

Province	Geotype	Removed at least once a week	Removed less often than once a week	Communal refuse dump	Own refuse dump	Dump or leave rubbish anywhere	Other
Western Cape	Rural	0,0	0,0	0,0	0,0	0,0	0,0
	Urban	95,5	0,3	2,8	1,1	0,3	0,0
	Metro	93,0	0,6	6,1	0,1	0,2	0,0
	Total	91,2	1,8	5,0	1,8	0,3	0,1
Eastern Cape	Rural	0,1	0,0	0,0	98,0	1,9	0,1
	Urban	75,1	4,2	1,0	16,9	0,7	2,2
	Metro	61,9	35,8	0,0	0,7	1,2	0,4
	Total	35,2	7,6	0,3	54,6	1,5	0,8
Northern Cape	Rural	0,0	0,0	1,6	94,6	3,7	0,0
	Urban	85,3	3,1	1,3	2,8	7,1	0,4
	Metro	0,0	0,0	0,0	0,0	0,0	0,0
	Total	69,8	2,4	1,7	18,8	6,6	0,7
Free State	Rural	0,8	0,0	1,6	73,5	24,2	0,0
	Urban	90,1	2,1	0,3	3,1	4,0	0,4
	Metro	0,0	0,0	0,0	0,0	0,0	0,0
	Total	76,1	1,8	0,8	14,0	6,9	0,3
KwaZulu-Natal	Rural	1,4	0,8	1,2	92,5	4,2	0,0
	Urban	72,4	4,4	0,0	21,3	1,4	0,5
	Metro	86,6	3,9	0,3	8,9	0,3	0,0
	Total	51,9	3,1	0,6	42,5	1,9	0,1
North West	Rural	32,5	3,8	0,9	59,0	3,8	0,0
	Urban	90,1	2,4	1,2	2,5	3,8	0,1
	Metro	0,0	0,0	0,0	0,0	0,0	0,0
	Total	58,1	2,9	1,1	32,8	5,1	0,0
Gauteng	Rural	0,0	0,0	0,0	0,0	0,0	0,0
	Urban	89,3	2,9	0,4	6,5	1,0	0,0
	Metro	90,9	0,8	2,0	4,4	1,7	0,2
	Total	90,1	1,0	1,8	5,2	1,7	0,2
Mpumalanga	Rural	7,4	2,4	0,4	84,1	5,6	0,2
	Urban	75,8	2,9	2,1	14,1	5,1	0,0
	Metro	0,0	0,0	0,0	0,0	0,0	0,0
	Total	38,2	2,7	1,9	51,0	6,1	0,1
Limpopo	Rural	4,9	0,8	0,6	87,7	5,7	0,3
	Urban	89,0	2,0	0,5	8,3	0,3	0,0
	Metro	0,0	0,0	0,0	0,0	0,0	0,0
	Total	20,2	1,3	0,9	71,1	6,3	0,3
South Africa	Rural	6,9	1,2	0,6	86,5	4,6	0,1
	Urban	84,3	2,8	1,0	9,0	2,6	0,4
	Metro	89,2	2,9	2,4	4,2	1,2	0,1
	Total	63,5	2,6	1,6	29,2	3,0	0,2

Table 3.3: Population statistics per province in South Africa (StatsSA, 2011 -2014).

Province	Census 1996	Census 2001	Census 2011	Mid-year population estimates 2013	Mid-year population estimates 2014	Number of consumer units receiving SWM services 2011	Number of consumer units receiving SWM services 2012	Number of consumer units receiving SWM services 2013	Number of consumer units receiving SWM services 2014
Eastern Cape	6 302 525	6 436 763	6 562 053	6 620 100	6 786 900	752 350	766 347	766 131	802 073
Free State	2 633 504	2 706 775	2 745 590	2 753 200	2 786 800	526 830	563 273	628 430	655 580
Gauteng	7 348 423	8 837 178	12 272 263	12 728 400	12 914 800	2 652 189	2 574 182	2 763 184	2 978 735
KwaZulu-Natal	8 417 021	9 426 017	10 267 300	10 456 900	10 694 400	2 077 319	1 442 130	1 423 290	1 466 259
Limpopo	4 929 368	5 273 642	5 404 868	5 518 000	5 630 500	363 391	384 973	412 282	425 127
Mpumalanga	2 800 711	3 122 990	4 039 939	4 128 000	4 229 300	405 734	420 509	513 075	549 468
North West	3 354 825	3 669 349	3 509 953	3 597 600	3 676 300	465 048	464 993	491 175	573 424
Northern Cape	840 321	822 727	1 145 861	1 162 900	1 166 700	211 094	215 811	221 478	227 012
Western Cape	3 956 875	4 524 335	5 822 734	6 016 900	6 116 300	1 257 378	1 176 365	1 172 648	1 184 892
South Africa	40 583 573	44 819 776	51 770 561	52 982 000	54 002 000	8 711 333	8 008 583	8 391 693	8 862 570

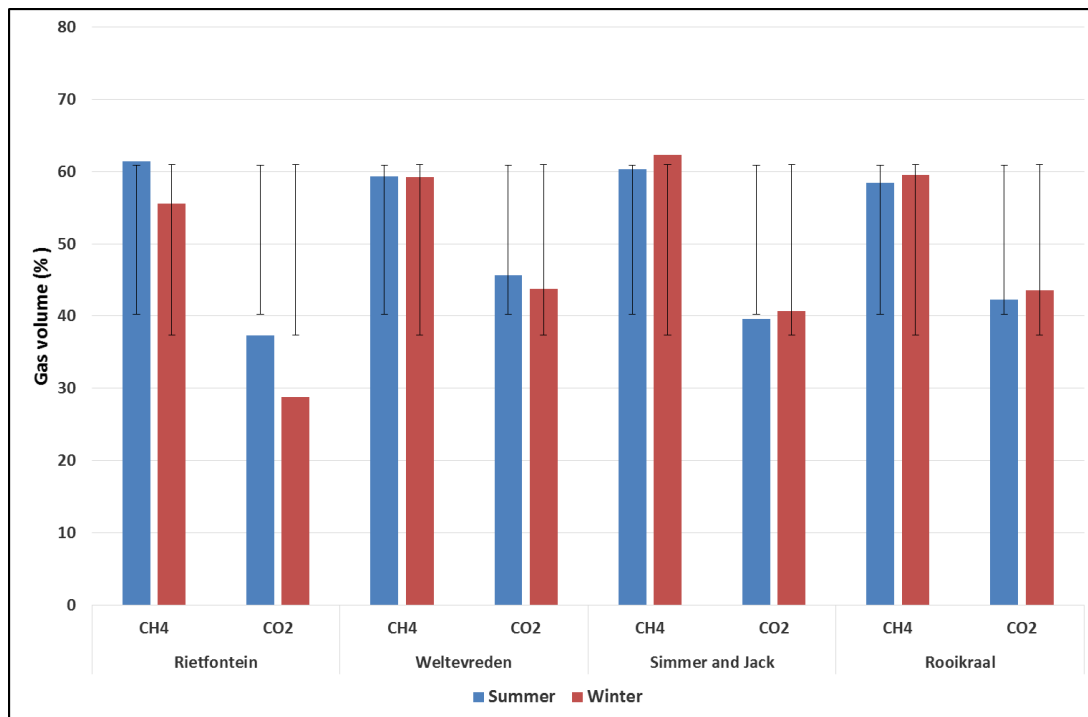
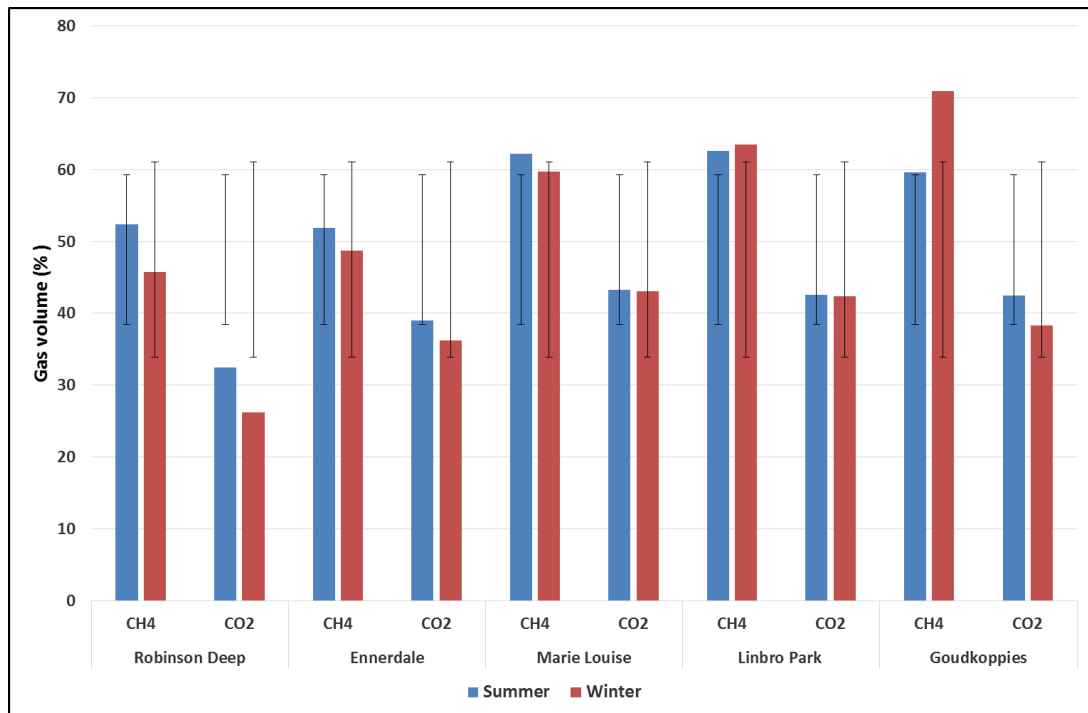


Figure 3.1: Average CH_4 and CO_2 (% v/v) measured from subsurface gas probe along the boundary of the landfill sites in City of Johannesburg (top) and Ekurhuleni Metro Municipality (bottom) during summer and winter from 2009 – 2011. Error bars are given as (+/- 1 standard deviation).

The Gauteng province (the smallest of South Africa's provinces) has the largest population and waste generation rates per capita within the country and hence has the highest CH₄ emissions (Figure 3.2). It also has a high rate of weekly collection and disposal at 91.1% which means most waste is disposed at landfill sites adding to overall CH₄ emissions.

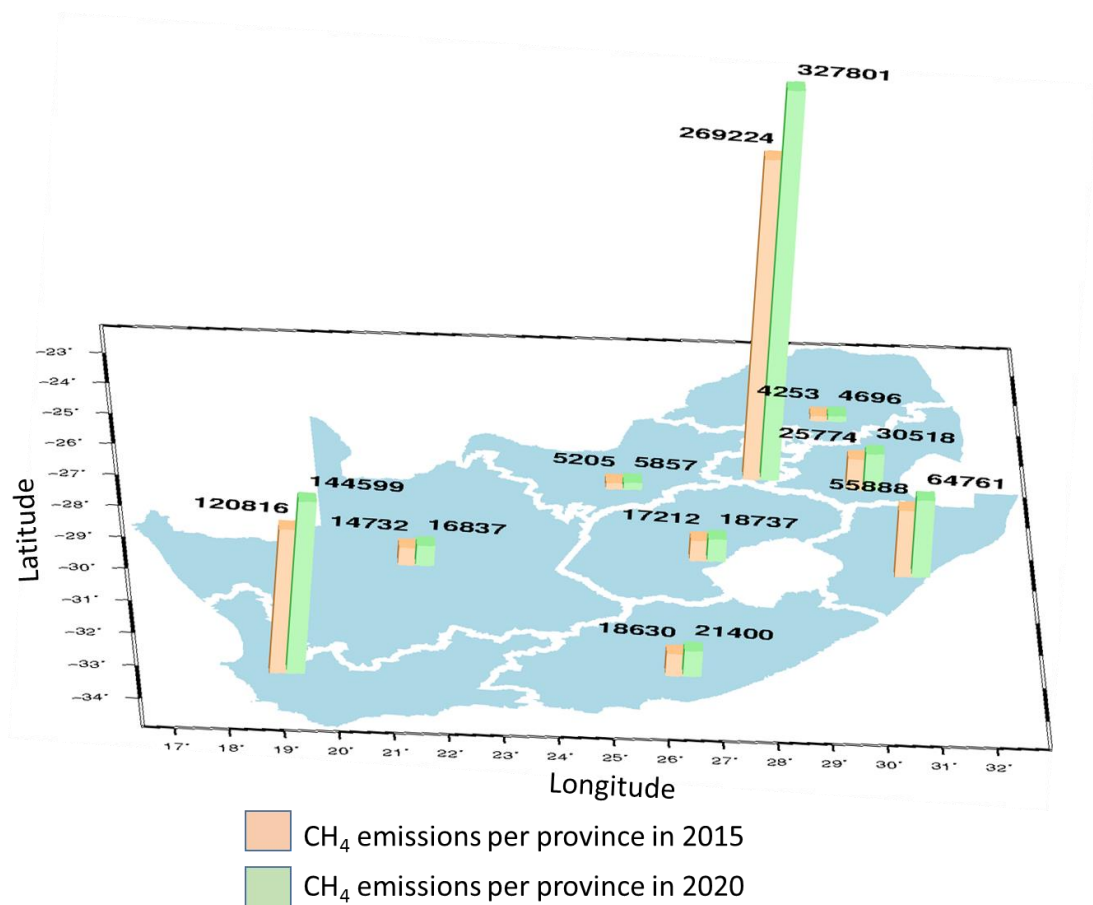


Figure 3.2: Tonnes of CH₄ generated per province in South Africa for 2015 and 2020 calculated in this study using IPCC Waste Model 2006.

The Western Cape is the fourth largest province in South Africa and has the lowest unemployment rate in the country. According to the National Waste Baseline Report (2012) it contributes 20% of all waste generated in the country. The Western Cape has higher CH₄ emissions than both Kwa-Zulu Natal and the Eastern Cape

despite these provinces having a larger population compared. This is mainly due to higher urbanisation rates within the Western Cape with larger rural areas found in the other two provinces. As a result, waste collection rates in the Western Cape at 93% far exceed both Kwa-Zulu Natal and the Eastern Cape at 42.8% and 55% respectively. In addition, the Eastern Cape is one of South Africa's poorest provinces with a waste generation rate of 0.31 kg/capita/day and only contributing 4% to overall waste generation in the country. Due to the subtropical climate of Kwa-Zulu Natal the province has higher waste degradation rates which increases its overall CH₄ emissions.

In total in South Africa 531 735 tCH₄/annum and 635 207 tCH₄/annum emission were predicted for 2015 and 2020 respectively. Using a GWP of 21 for CH₄ this equates to 11 166 431 tCO₂e and 13 293 370 tCO₂e for 2015 and 2020 respectively and a GWP of 25 gives total emissions at 13 339 350 tCO₂e and 15 880 178 tCO₂e. A GWP of 21 will be used in this study as all projects have used GWP of 21 for the first commitment period of the Kyoto Protocol. Gauteng contributes 49.37% of all CH₄ emissions in South Africa in 2015. According to DEA's National Greenhouse Gas Inventory in 2010 the total GHG emissions in South Africa were estimated at 579 256 000 tCO₂eq with MSW contributing 15 535 000 tCO₂eq (contributing 2.6% to overall emissions). This study shows that the contribution to greenhouse gas emissions from the disposal of MSW is lower at 2.3%. Figure 3.2 highlights the major economic and social disparities between provinces in South Africa which affect each provinces contribution to the CH₄ emissions from waste disposal. This is further highlighted by the Metro Municipalities within the provinces. Three Metros are within Gauteng, two in the Eastern Cape one in the Western, one in

Kwa-Zulu Natal and one in the Free State. Population statistics for the Metros were used as given in Table 3.4. The City of Johannesburg (CoJ) has the highest population as well as highest weekly waste collection rates at 95.3%.

Table 3.4: Population statistics for the eight major Metro Municipalities in South Africa (StatsSA, 2001 – 2014).

Metro	Population (2001)	Percentage of weekly refuse removal 2001 (%)	Population (2011)	Number of Households (2011)	Percentage of weekly refuse removal 2011
City of Johannesburg	3 226 055	90.9	4 434 827	1 434 856	95.3
City of Cape Town	2 892 243	94.3	3 740 026	1 068 573	94.3
eThekweni	3 090 122	85.7	3 442 361	956 713	86.1
Ekurhuleni	2 481 762	87.9	3 178 470	1 015 465	88.4
City of Tshwane	2 142 322	75.2	2 921 488	911 536	80.7
Nelson Mandela Bay	1 005 779	86.1	1 152 115	324 292	82.9
Buffalo City	704 855	71.2	755 200	223 568	70.4
Mangaung	645 440	60	747 431	231 921	78.9

Mean waste generation rates per capita were used according to different income groups – low, middle and high (Table 3.5).

Table 3.5: Waste generated per capita determined by waste characterisation studies conducted in City of Johannesburg and South Africa

Income Level	Waste generation kg/capita/day				
	DEAT (2006)	Fiehn and Ball (2005)	CoJ (Sept 2014)	CoJ (Nov 2014)	Mean
Low	0.41	0.41	0.99	0.70	0.555
Medium	0.74	0.74	0.91	0.92	0.825
High	1.29	1.29	1.22	1.26	1.275

Previous waste characterisation studies conducted with the City of Johannesburg showed that low income households contribute 52% of the waste generated, middle

income 21% and high income households 27% in the summer (CoJ, 2014). Although this changes slightly during winter with middle income households exceeding high income. This is thought to be mainly due to the garden waste contribution during the summer by high income households (CoJ, 2014). The highest CH₄ emissions are from City of Johannesburg contributing to over 60 000 tCH₄/annum in 2015 and exceeding 70 000 tCH₄/annum in 2020 (Figure 3.3). Ekurhuleni (EMM) and City of Tshwane (CoT) are fourth and fifth highest respectively. All three (CoJ, EMM and CoT) Metro Municipalities are within the Gauteng province.

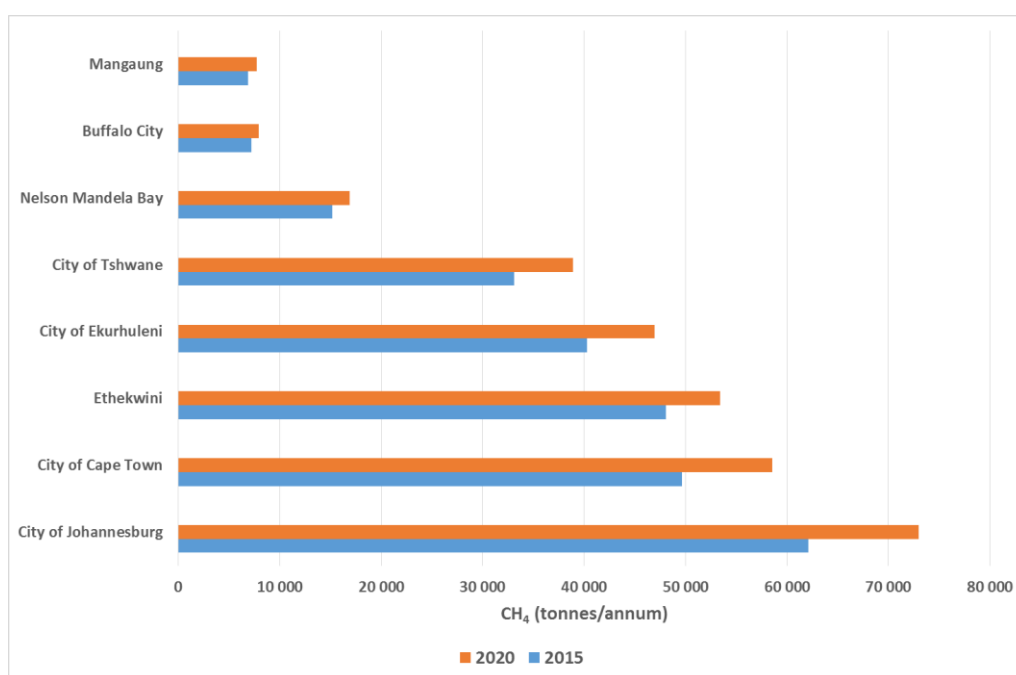


Figure 3.3: Tonnes of CH₄ generated within 8 Major Metro Municipalities in South Africa for 2015 and 2020 as modelled using IPCC Waste Model 2006.

The City of Cape Town (CCT) has slightly higher emissions than eThekweni which can be attributed to the higher waste collection rates in Cape Town at 94.3% compared to 86.1% within eThekweni. In addition, the subtropical climate and higher degradation rates also add to the predicted CH₄ emissions for eThekweni.

In total the Metro Municipalities contribute 50.64% of total emissions in South Africa for 2015 and 52.25% in 2020 (262 480 tCH₄/annum and 303 317 tCH₄/annum) respectively. Using a GWP of 21 for CH₄ this equates to 5 512 080 tCO₂e and 6 369 661 tCO₂e for 2015 and 2020 respectively, with 2 846 170 tCO₂e emissions from the three Metro Municipalities in Gauteng province only (i.e. City of Johannesburg, City of Tshwane and Ekurhuleni Metro Municipality). Gauteng accounts for 48.36% of all CH₄ emissions from waste disposal amongst the eight Metro Municipalities. The results, showing that Metro Municipalities make a significant contribution to emissions, are indicative of the vast disparities in waste disposal between urban and non-urban areas in the country. Vanyaza (2014) modelled emissions from seven municipalities in South Africa and determined the total emissions from MSW to be between 5 000 000 tCO₂e in 2015 and 6 500 000 tCO₂e in 2020.

To assess the contribution of individual landfill sites to the predicted CH₄ emissions and to investigate exactness of the IPCC Waste Model in calculating emissions the LandGEM and GasSim LFG estimation models were used in comparison. These models allow for individual landfill sites to be modelled and are based on waste tonnages accepted at each site from opening to closure. CH₄ emissions at four landfill sites in the CoJ and four within EMM were predicted. Tonnages are recorded by weighbridges at the site entrance. When weighbridges are not functional waste volumes are determined by the number of vehicles entering the site and associated volumes calculated thus increasing levels of uncertainty. Weighbridges within CoJ were often not functional whilst EMM weighbridges were fully functional throughout the year with minimal downtime for maintenance.

Thus, tonnages from EMM are fairly accurate and have a high level of confidence. Exact volumes from 2001 were available and tonnages were projected backward (using an average 2% growth rate) to open year for each landfill site as required by the models (Figure 3.4).

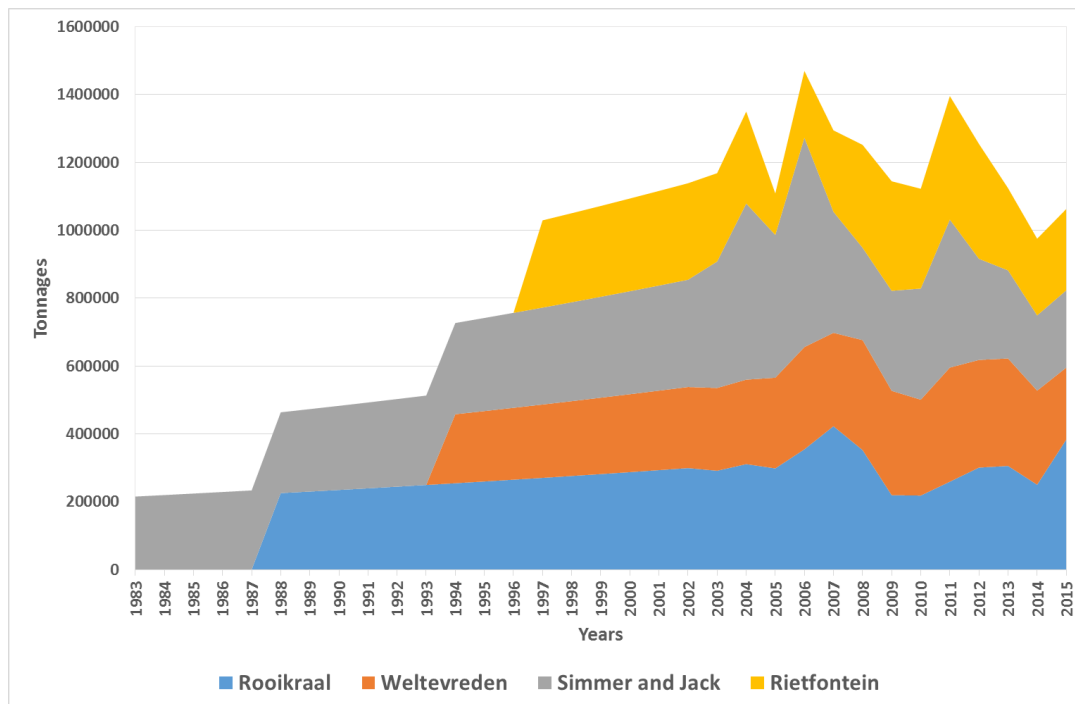


Figure 3.4: Waste tonnages accepted at Ekurhuleni Metro Municipality landfill sites.

Default waste degradation rates were used for all models. GasSim assigns degradation rates for dry, average and wet climates and further breaks it down into slow, moderate or rapid decomposition rates based on different waste fractions (Table 3.6). For example, in a dry climate, garden and food waste decompose at high rates once disposed at a landfill site and have 100% rapid degradation applying a CH_4 generation rate constant k of 0.076 whilst textiles would decompose slowly at a k value of 0.013. Whereas only 75% of the paper fraction of the waste will degrade slowly at a k of 0.013 (in a dry climate) and the remaining 25% will degrade

moderately at a k of 0.046 year^{-1} (in a dry climate). This differs to LandGEM which assumes a single degradation rate ($k = 0.04$ in this case) and IPCC which assumes 100% degradation of individual waste fractions at a given k value. Higher k values result in higher CH_4 generation from the landfill.

Table 3.6: CH_4 generation rate constant k (year^{-1}) values used for GasSim. Degradation rates are provided for slow, moderate and rapidly degrading waste per fraction.

GasSim	Dry	Average	Wet
Slow	0.013	0.046	0.076
Moderate	0.046	0.076	0.116
Rapid	0.076	0.116	0.694

	Slow (%)	Moderate (%)	Rapid (%)
Paper	75	25	0
Wood	25	75	0
Textiles	100	0	0
Disposable Nappies	0	100	0
Garden waste	0	0	100
Food	0	0	100
Plastics/non degradable	100	0	0

Predicted CH_4 generation for all landfill sites is shown in Figure 3.5 and 3.6. Predicted CH_4 generation using LandGEM is higher than both IPCC and GasSim. This is attributed to the model applying a single k value for all waste degradation and that the waste volumes entered are not separated into different waste compositions. Nevertheless, the CH_4 generation using the IPCC Waste Model is only slightly lower than LandGEM. This is expected as despite the waste being divided into fractions with assigned k values in the IPCC Waste Model, the average k is 0.045 for all waste fractions using the default k values in the model.

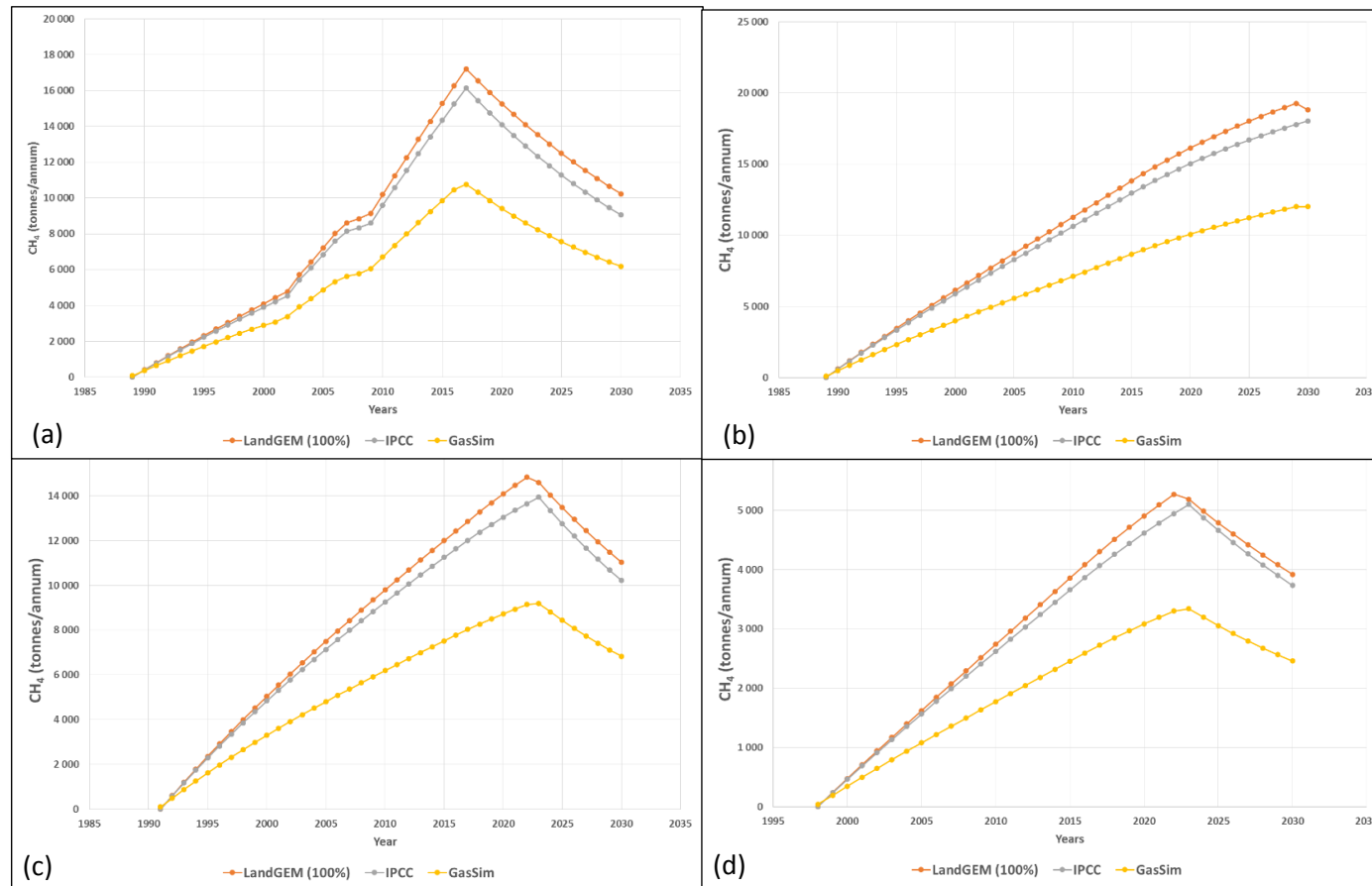


Figure 3.5: Comparison of CH₄ emissions per annum from landfill open year to 2030 using LFG estimation models (LandGEM, IPCC Waste Model and GasSim) for Robinson Deep (a), Goudkoppies (b), Marie Louise (c) and Ennerdale (d) landfill sites.

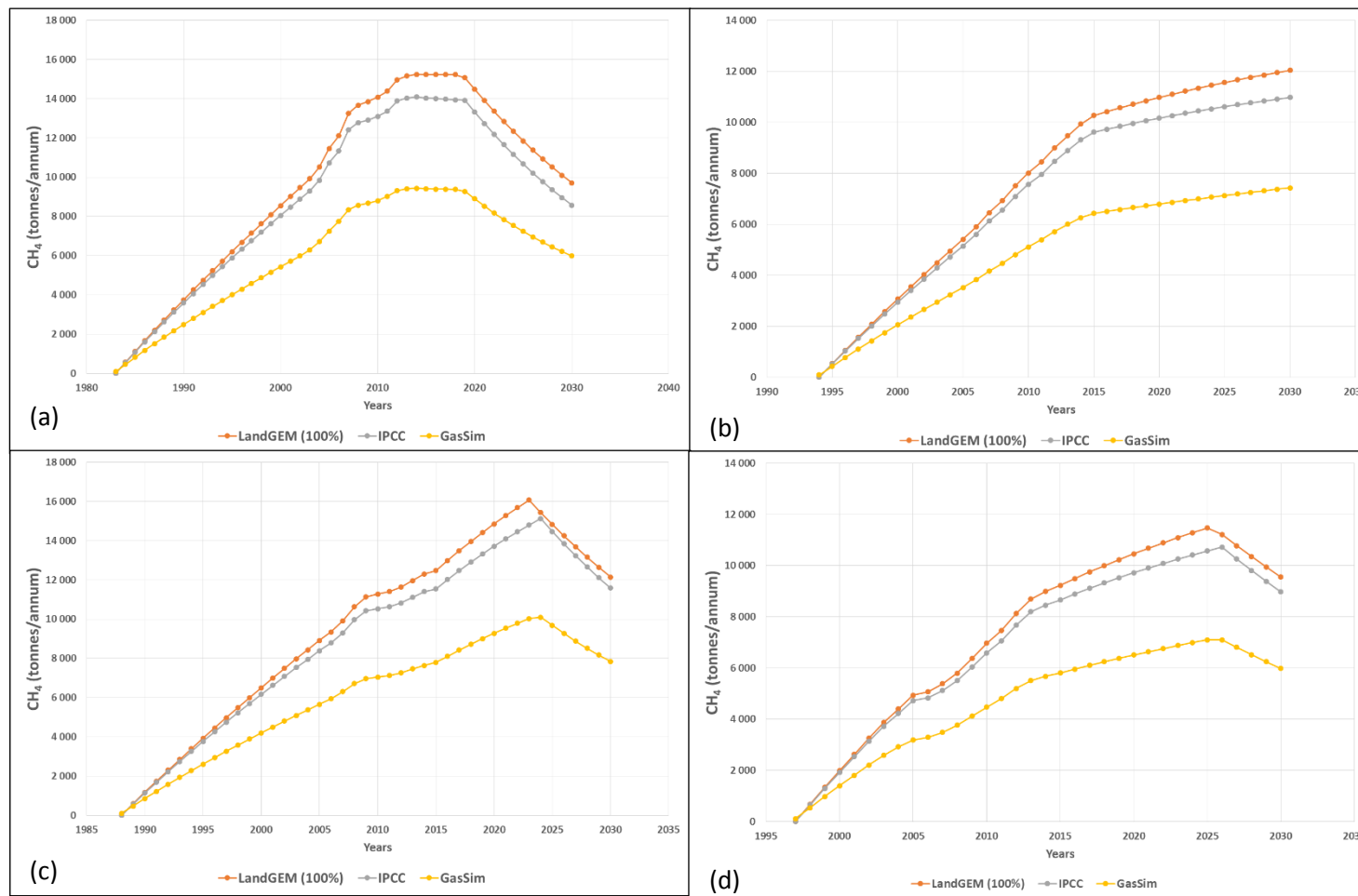


Figure 3.6: Comparison of CH₄ emissions per annum from landfill open year to 2030 using LFG estimation models (LandGEM, IPCC Waste Model and GasSim) for Simmer and Jack (a), Weltevreden (b), Rooikraal (c) and Rietfontein (d) landfill sites.

LFG estimations compared to actual gas yields

To assess the applicability and exactness of the LFG estimation models in predicting CH₄ generation from landfills in South Africa, the estimated CH₄ generation was compared to actual gas yields from gas extraction systems. Six of the eight landfill sites used in the study have gas extraction systems installed at the sites with vertical and horizontal wells installed in different phases (Table 3.7).

Table 3.7: Vertical and horizontal well distribution at all landfill sites.

Landfill	Horizontal wells	Vertical wells
Robinson Deep		
Cell 1, 2, 3	4	65
Marie Louise		
Cell 1	0	28
Rooikraal		
Stage 1	0	36
Stage 2 A and B	15	0
Simmer and Jack		
Cell 2, 3 and 4	3	38
Cell 5 and 6	10	12
Weltevreden		
Cell 1, 2 and 3	4	22
Cell 4 and 5	17	0
Rietfontein		
Cell 1, 2 and 3	0	25

It is also important to note that the coverage of the extraction system is not 100% at all sites. In fact, the highest percentage of coverage is Simmer and Jack at 68% (Table 3.8) and the landfill has reached capacity for well installations. Hence, taking that into consideration current LFG yields were projected to an assumed maximum coverage at 70% to for all sites to assess model functionality. An average of the gas yields predicted by LandGEM, IPCC Waste Model and GasSim was calculated for each year (relevant to each landfill site) and compared to actual gas yields.

Table 3.8: Percentage coverage of gas extraction system at landfill sites.

	Total Landfill Footprint	Current Footprint In Use	Gas Extraction Area	Coverage of Gas Extraction (Total Footprint)	Coverage of Gas Extraction (Current Footprint)
Weltevreden	1 130 373	585 918	272 384	24%	46%
Rooikraal	1 007 214	644 578	351 331	35%	55%
Rietfontein	451 127	306 095	121 216	27%	40%
Simmer and Jack	450 382	450 382	305 834	68%	68%

Average LFG flow rates at Robinson Deep were above 1000 m³/hr and gradually decreased to 500 m³/hr between May 2013 and May 2015. This decrease could be attributed to the initial residual gas build up extracted from the site as the gas management system was commissioned. LFG yields at current 35% coverage is well below predicted average gas flows for the same period for LandGEM, IPCC Waste Model and GasSim. However, considering a 70% coverage LFG yields are above CH₄ generation predicted by GasSim (Figure 3.7, top).

Marie Louise actual yields average below 500 m³/hr at 25% coverage. Even at projected 70% coverage monitored gas flows are well below predicted rates by LFG estimation models but falls closely to predicted yields by GasSim (Figure 3.7, bottom).

Gas wells are progressively installed at EMM landfill sites as waste is disposed and sufficient for new installations. All vertical and horizontal wells have been commissioned at the Simmer and Jack landfill site and the landfill has reached capacity at 70% coverage. LFG yields have increased over the years peaking at 2000 m³/hr. As newly installed individual wells are opened and gas extracted, it adds to the overall gas yielded at the site. The landfill is also reaching closure

forecasted in 2018 and therefore it is expected to be at, or close to, its maximum LFG generation potential. LFG yields between September 2014 and April 2015 are above CH₄ generation predicted by GasSim (Figure 3.8, top).

Actual LFG yields at the Weltevreden landfill site (Figure 3.8, bottom) are well below predicted generation for all models at current 50% coverage and projected 70% coverage. CH₄ yields peak at 800 m³/hr and average at 500 m³/hr. Actual gas yields at 55% coverage and projected 70% coverage for Rooikraal are above GasSim predicted yields from July 2014 (Figure 3.9, top) this is attributed to the commissioning of newly installed wells at the site resulting in higher gas yields. Rietfontein is well below all predicted flow rates (Figure 3.9, bottom).

The LandGEM and IPCC Waste Model over estimate CH₄ generation at all sites whilst GasSim is the closest estimation to actual gas yields at the sites. Since the GasSim model takes into consideration the differing decomposition rates for waste fractions, that is, it allows for certain percentages of the waste stream to decay at a later stage, it more accurately predicts CH₄ generation through the life of the waste disposed at the landfill sites. Thompson *et al.*, (2009) compared estimated CH₄ generation (using German EPER, TNO, Belgium, LandGEM, and Scholl Canyon models) to actual gas yields at 35 landfill sites in Canada. The study showed that The Belgium, Scholl Canyon and LandGEM models produced the best results of the existing models with respective mean absolute errors compared to methane generation rates of 91%, 71%, and 89% at 0.50 DOC_f and 171%, 115%, and 81% at 0.77 DOC_f. LFG estimation models should be used for regulatory purposes and to provide an indication of the potential for LFG emissions.

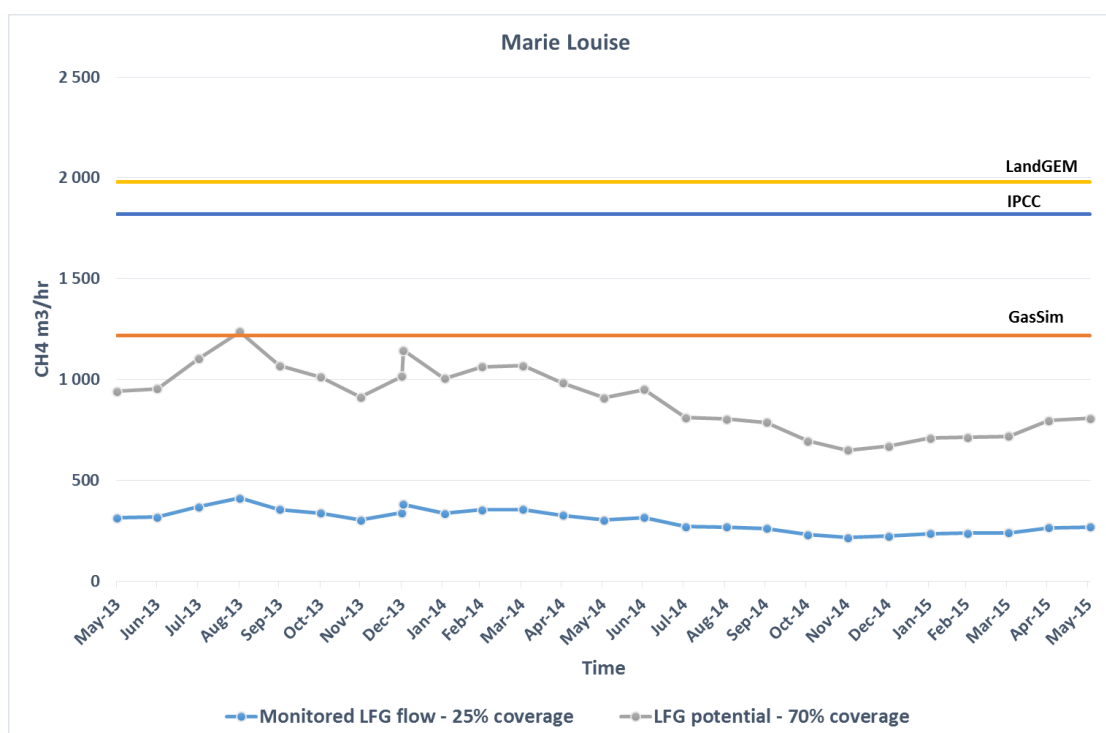
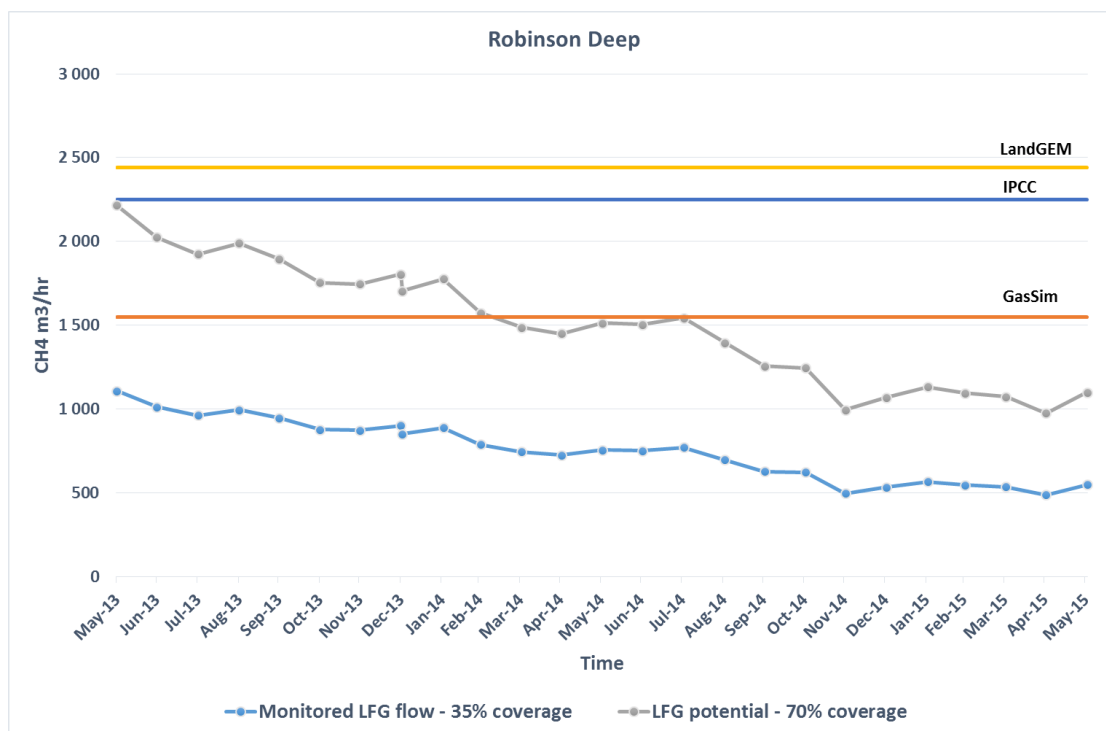


Figure 3.7: Actual LFG yields compared to predicted CH₄ generation for Robinson Deep (top) and Marie Louise (bottom)

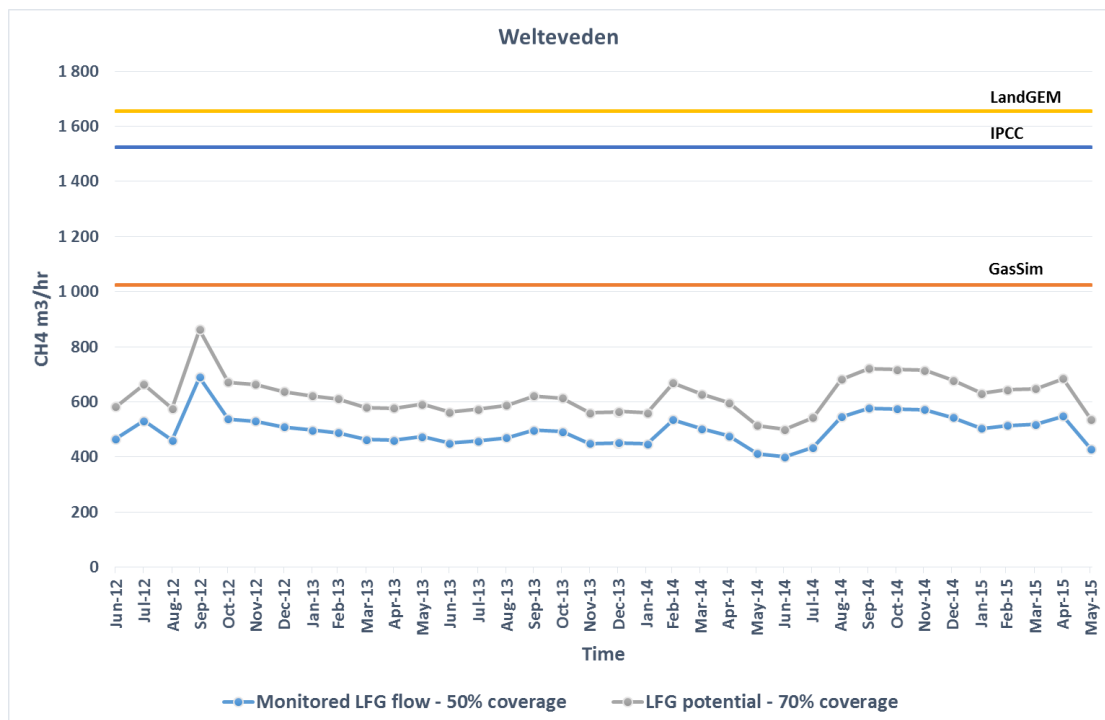
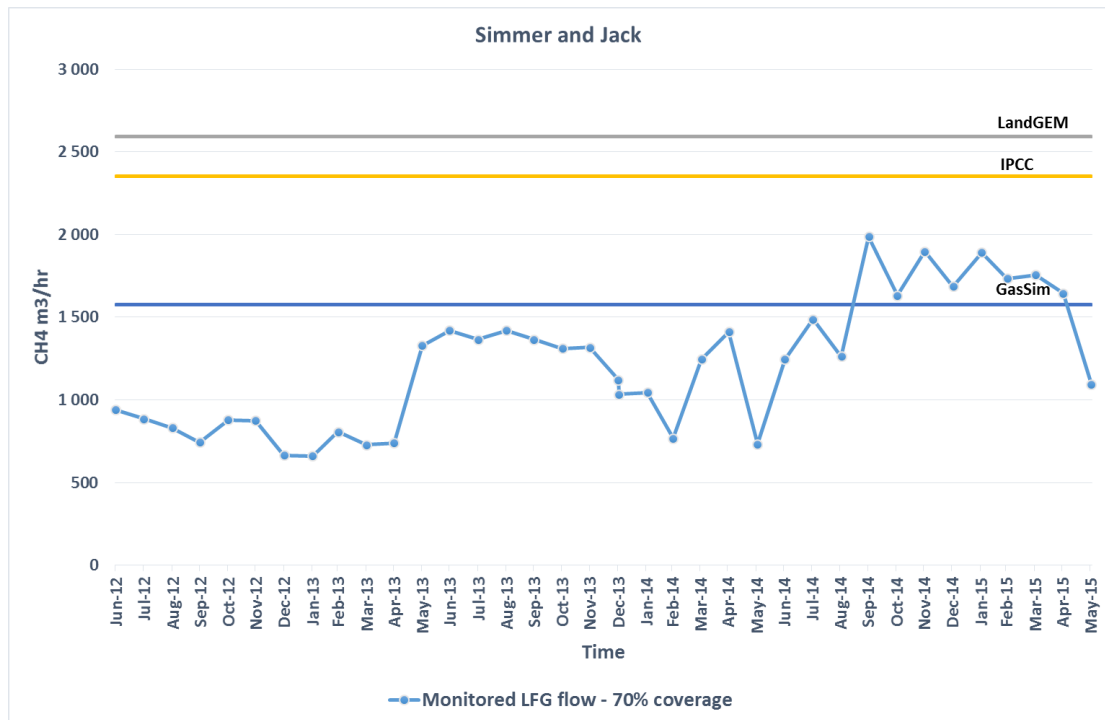


Figure 3.8: Actual LFG yields compared to predicted CH₄ generation for Simmer and Jack (top) and Weltevreden (bottom)

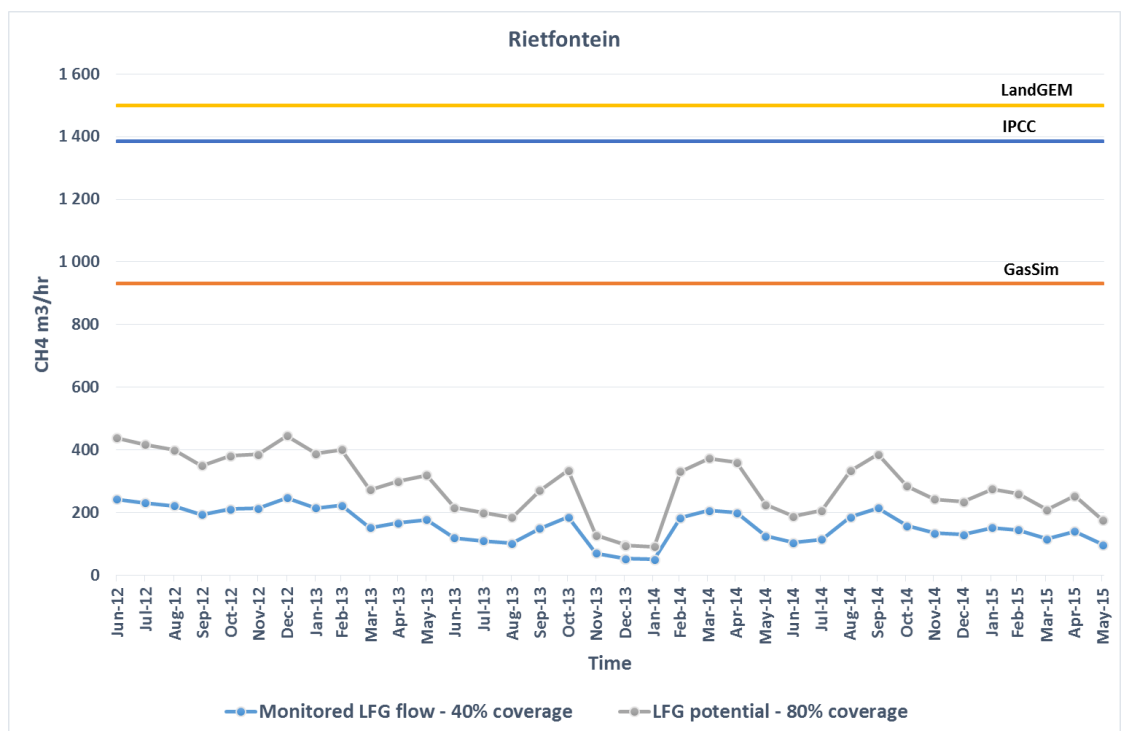
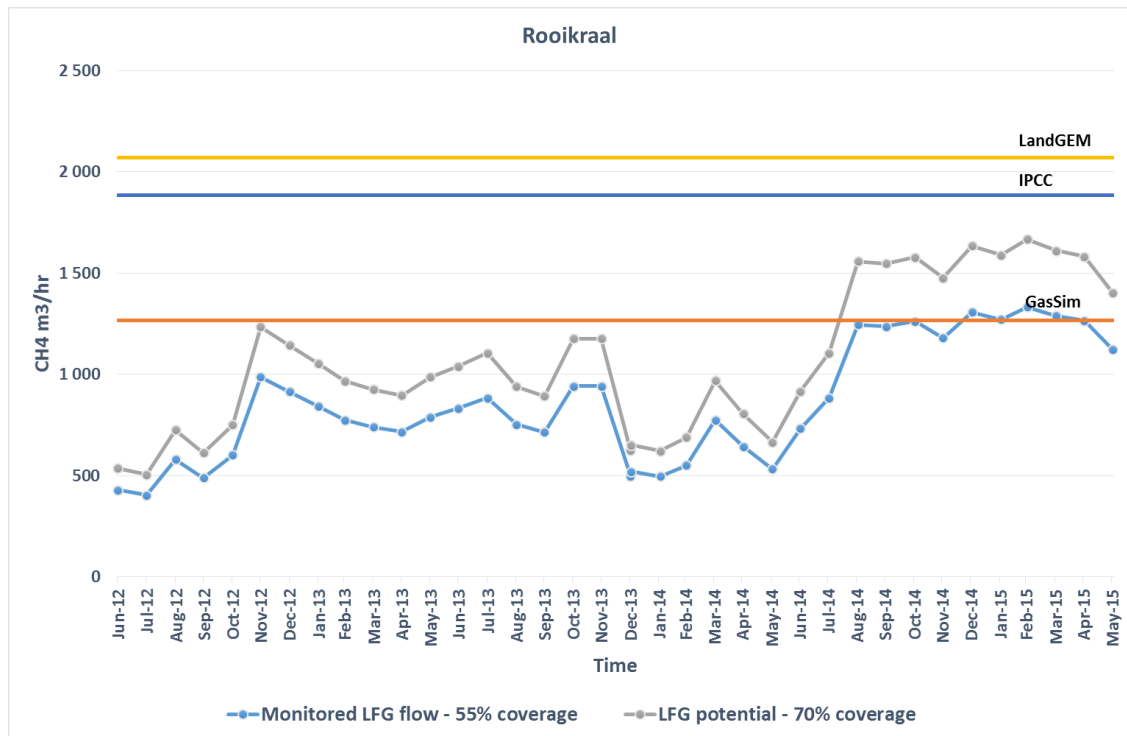


Figure 3.9: Actual LFG yields compared to predicted CH₄ generation for Rooikraal (top) and Rietfontein (bottom).

Direct emissions from the surface landfill sites

Direct emissions from the landfill surface at Robinson Deep and Marie Louise were determined using flux chambers placed at specific points across the landfill surface (as shown in Figure 2.11, in Chapter 2). CH₄ and CO₂ emissions were calculated at each site. All measurements were on landfill cover soils with no vegetation cover present. CH₄ and CO₂ emissions were generally low at both landfill sites, however emissions were higher at Marie Louise than Robinson Deep. CH₄ emissions at Robinson Deep peak at 0.17 g/m²/day (Figure 3.10, (a)) whilst the highest CH₄ emission measured at Marie Louise was 0.44 g/m²/day. These emissions are low compared to previous studies. Chanton and Liptay (2000) measured CH₄ emissions between 200 g/m²/day through a mulch soil cover and 9000 g/m²/day through a clay cover. Fourie and Morris (2004) measured CH₄ and CO₂ fluxes at four landfill sites in South Africa. Measured fluxes in the study ranged between -45 g/m²/day and 638 g/m²/day during winter at an operational landfill site. Mean fluxes measured by Fourie and Morris (2004) ranged between 12.7 g/m²/day and 56.5 g/m²/day, higher than fluxes measured during this study. However, Bogner and Matthews (2003) using a limited number of whole landfill CH₄ emissions measurements in Europe, the United States, and South Africa show CH₄ emissions variations from 0.03 to 0.3 g/m²/day.

Random sample CH₄ emission fluxes range between -0.0024 g/m²/day and 0.0338 g/m²/day, highlighting the spatial variability of CH₄ emissions across the landfill surface. Bogner *et al.*, (2011) show CH₄ emissions can vary over 3-4 orders of magnitude during a single field campaign. Emissions can vary due to differences in

waste amounts, age and composition in different parts of the landfill, hence emissions vary greatly over the surface over a few meters as described by Farquhar and Rovers (1973), Christensen and Kjeldsen (1996) and Morris (2001).

The low CH₄ and CO₂ surface fluxes could be attributed to the slow degradation of waste at the surface of the active cell. The waste degrades aerobically and reaches methanogenesis at a later stage or once another layer of waste is disposed above. High oxidation rates of CH₄ in cover soils to CO₂ also result in lower emissions to the atmosphere. Methanogenesis is reached within the landfill site as shown by the LFG extraction rates. A study measuring CH₄ and CO₂ fluxes at a municipal landfill site in Argentina found no CH₄ emissions and CO₂ emissions between 5 g/m²/day to 214 g/m²/day (in 50 sample points) and 31 g/m²/day to 331 g/m²/day (at 107 sampling points). The study found that the shallow waste placement and insufficient compaction of the waste resulted in aerobic waste decomposition and that waste degradation underwent anaerobic oxidation and never reached methanogenesis (Sanci and Panarello, 2012). CO₂ emissions are higher than CH₄ emissions at both sites ranging between -2.87 g/m²/day and 13.27 g/m²/day at Robinson Deep (Figure 3.10, (b)). This compares well with Fourie and Morris (2004) which also found that CO₂ emissions were higher than CH₄ emissions at all four sites used in the study. The higher CO₂ emissions relative to CH₄ emissions could be attributed to CH₄ oxidation in cover soils, however this was not determined in this study. Previous studies have shown that oxidation of CH₄ at some landfill sites can be between 30 and 100% (Bogner *et al.*, 1995; Mancinelli, 1995; Czepiel *et al.*, 1996a; Whalen *et al.*, 1990; Christopherson *et al.*, 2000).

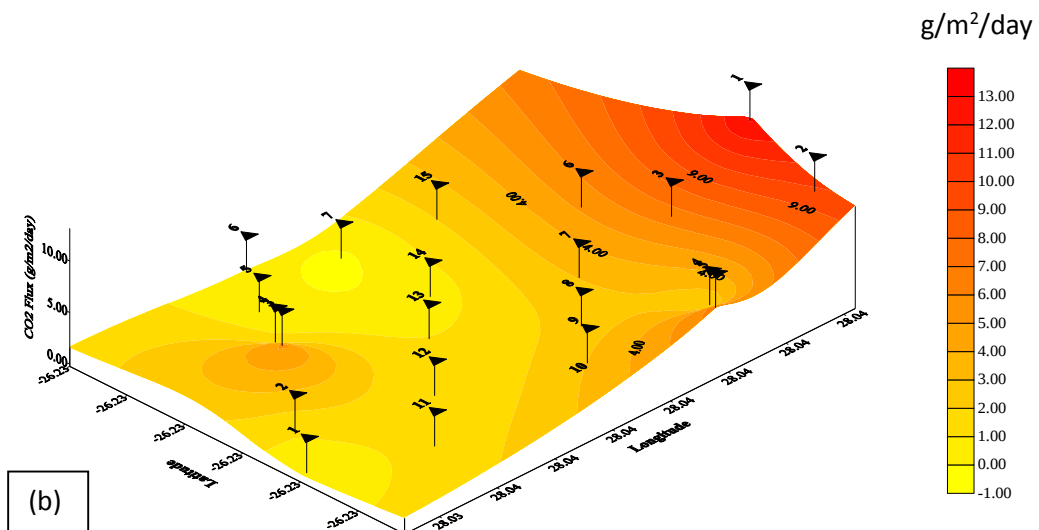
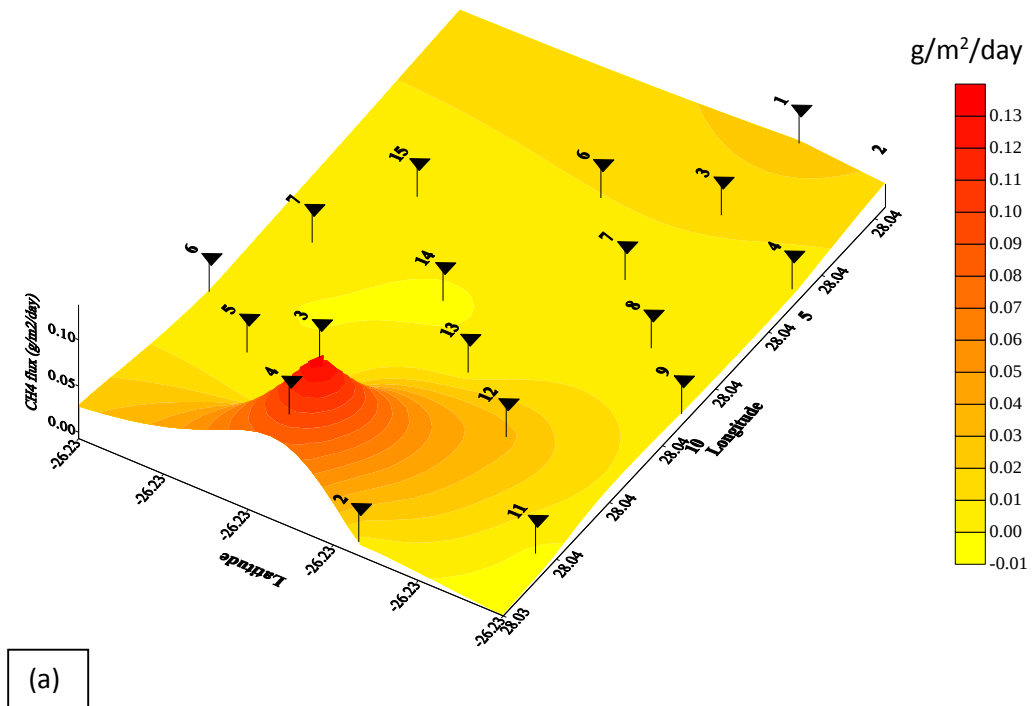


Figure 3.10: CH₄ (a) and CO₂ (b) flux emissions at Robinson Deep landfill site. Flux chamber locations are highlighted and numbered using black triangles.

Flux emission rates measured during the passing of a cold front were higher than under normal climatic conditions. Maximum flux was measured at 0.2579 g/m²/day

at Robinson Deep during a passing cold front (Table 3.9). This is consistent with previous studies which measured landfill CH₄ emissions during a passing cold front. Cziepel *et al.*, (2003) show that a barometric pressure decrease of 10 hPa causes a tripling of the CH₄ emissions from a landfill. Atmospheric pressure measurements taken show an average decrease of approximately 8 mbar during a cold front. Thus resulting in an increase in pressure gradient between CH₄ within the landfill and the atmosphere.

Table 3.9: Overall CH₄ emission fluxes measured at Robinson Deep and Marie Louise landfill sites.

	Robinson Deep	Cold front measurements at Robinson Deep	Marie Louise
Sampling points	22	8	6
Atm pressure (mbar)	838 – 841	830 - 833	838 - 841
Temp (Degrees Celsius)	12 – 18	5 - 20	7 - 11
Vegetation cover	None	none	none
CH₄ emissions (g/m²/day)			
Range	-0.0024 – 0.1762	0.0007 – 0.2579	0.0005 – 0.4414
Std dev	0.0527	0.1436	0.1727
Mean	0.0242	0.0925	0.1084
CO₂ emissions (g/m²/day)			
Range	-2.8720 – 13.2757	0.3159 – 6.5732	-0.6084 – 22.3700
Std dev	3.0772	3.0267	8.0377
Mean	2.1965	2.0413	5.0067

Flux measurements are lower when compared to CALMIM emissions estimation model. CALMIM does not allow for negative fluxes and therefore negative fluxes measured lie outside the predicted emissions by the model. The model was run using default and site specific weather data (maximum and minimum temperature and rainfall) (Figure 3.11). It shows the warmer, wetter summer months and colder drier winter months. Asadi *et al.*, (2013) measured surface emission fluxes from

landfills in Australia and found similar patterns. CALMIM modelled high surface CH₄ emissions in winter. Gas recovery was included in the model run at 10% since the extraction system covered approximately 10% of the landfill at the time.

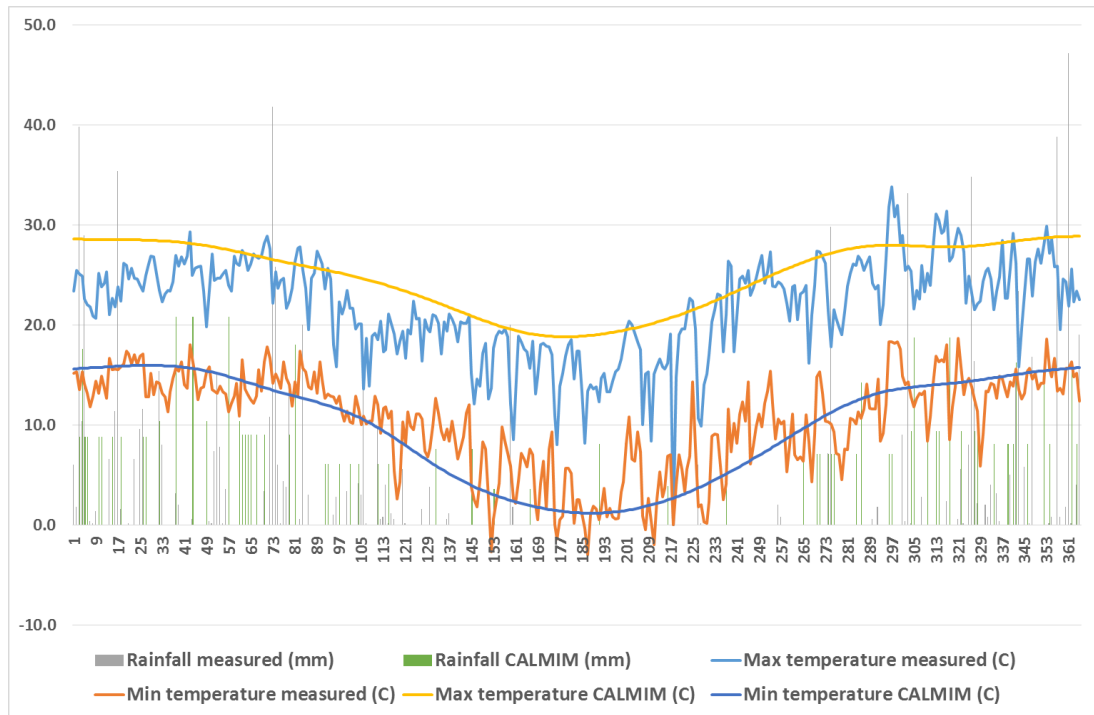


Figure 3.11: Measured meteorological data from the OR Tambo International airport compared to modelled data used by CALMIM.

Emissions estimations from CALMIM at the Robinson Deep landfill site, using default weather data, vary between 1.14 g/m²/day and 2.28 g/m²/day for daily cover, 72.61 g/m²/day and 130.23 g/m²/day for intermediate cover and 3.73 g/m²/day and 59.94 g/m²/day for final cover soils. Using site specific weather data the emissions predicted do not vary much from emissions using default weather data and are between 1.23 g/m²/day and 1.98 g/m²/day for daily cover, 55.91 g/m²/day and 130.33 g/m²/day for intermediate cover and 5.52 g/m²/day and 55.92 g/m²/day for final cover soils (Figure 3.11).

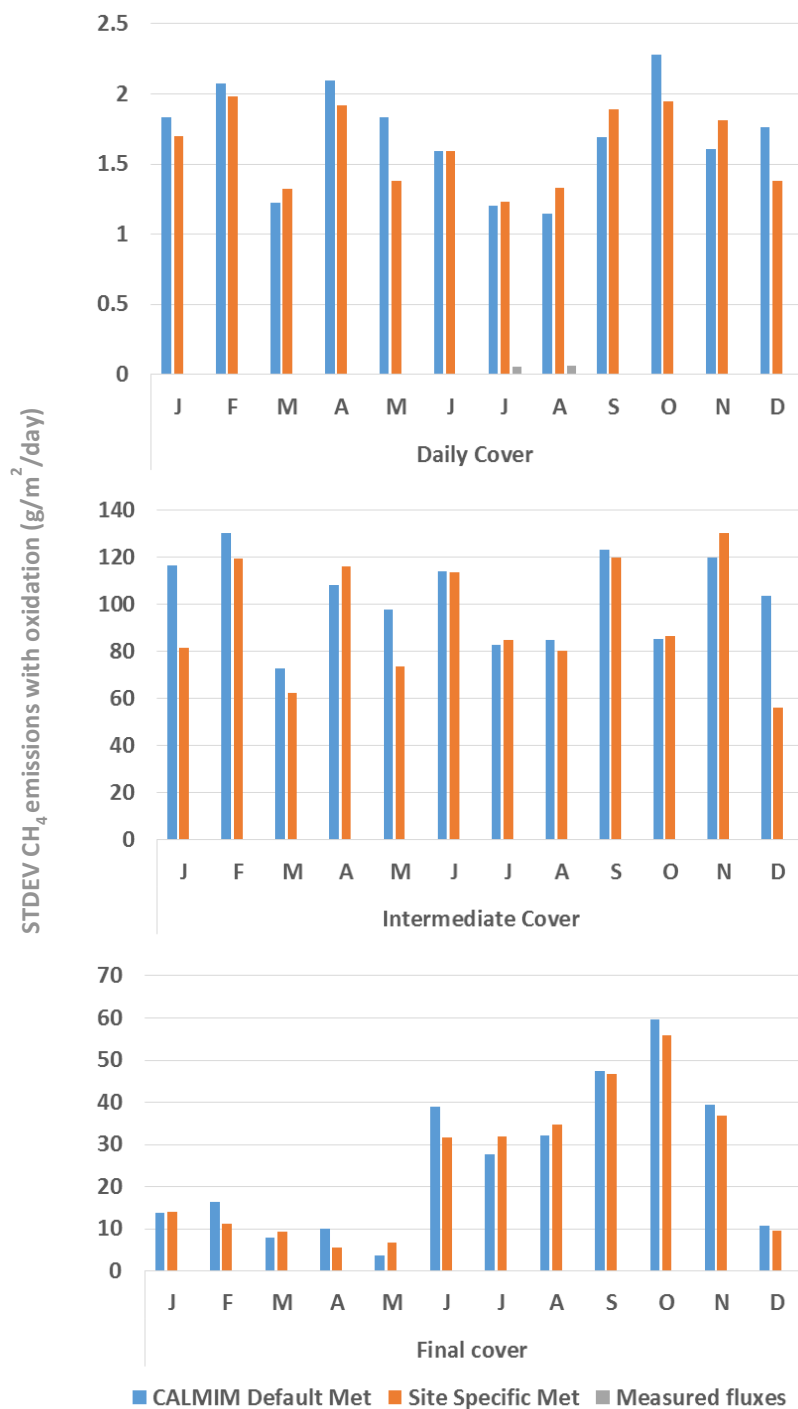


Figure 3.12: Modelled flux monthly emissions using CALMIM compared to measured flux emissions at the Robinson Deep landfill site. Measured emissions (in July and August) are comparable to emissions associated with daily cover and are much lower compared to intermediate and final cover soils.

Emissions measured at the surface of the landfill are comparable to emissions associated with daily cover soils. Intermediate and final cover soil emissions are

much higher. Standard deviation for fluxes measured at the landfill site was 0.0600 g/m²/day for the month of August and the modelled emission fluxes are 1.13 g/m²/day.

Emissions estimated using default weather data at the Marie Louise landfill site vary between 1.15 g/m²/day and 2.184 g/m²/day for daily cover, 73.71 g/m²/day and 132.45 g/m²/day for intermediate cover and 4.99 g/m²/day and 48.68 g/m²/day for final cover soils. Using site specific weather data the emissions range between 0.93 g/m²/day and 2.25 g/m²/day for daily cover, 36.82 g/m²/day and 151.42 g/m²/day for intermediate cover and 2.16 g/m²/day and 48.44 g/m²/day for final cover soils (Figure 3.12). Gas management was not included in the model run as an extraction system was not present at the site at the time flux measurements were taken.

The model shows high seasonal variability in surface emissions with oxidation and without. According to Bogner *et al.*, (2014) seasonal variability of CH₄ emissions can extend 2-3 orders of magnitude over an individual cover material at a specific site. This is highlighted by CALMIM where there are higher emissions in winter and lower emissions during the summer with oxidation. In total, modelled emissions from the landfill are 16 times higher (using daily cover soil) at 235845 kg/year compared to measured emissions from the surface equating to 14237 kg/year.

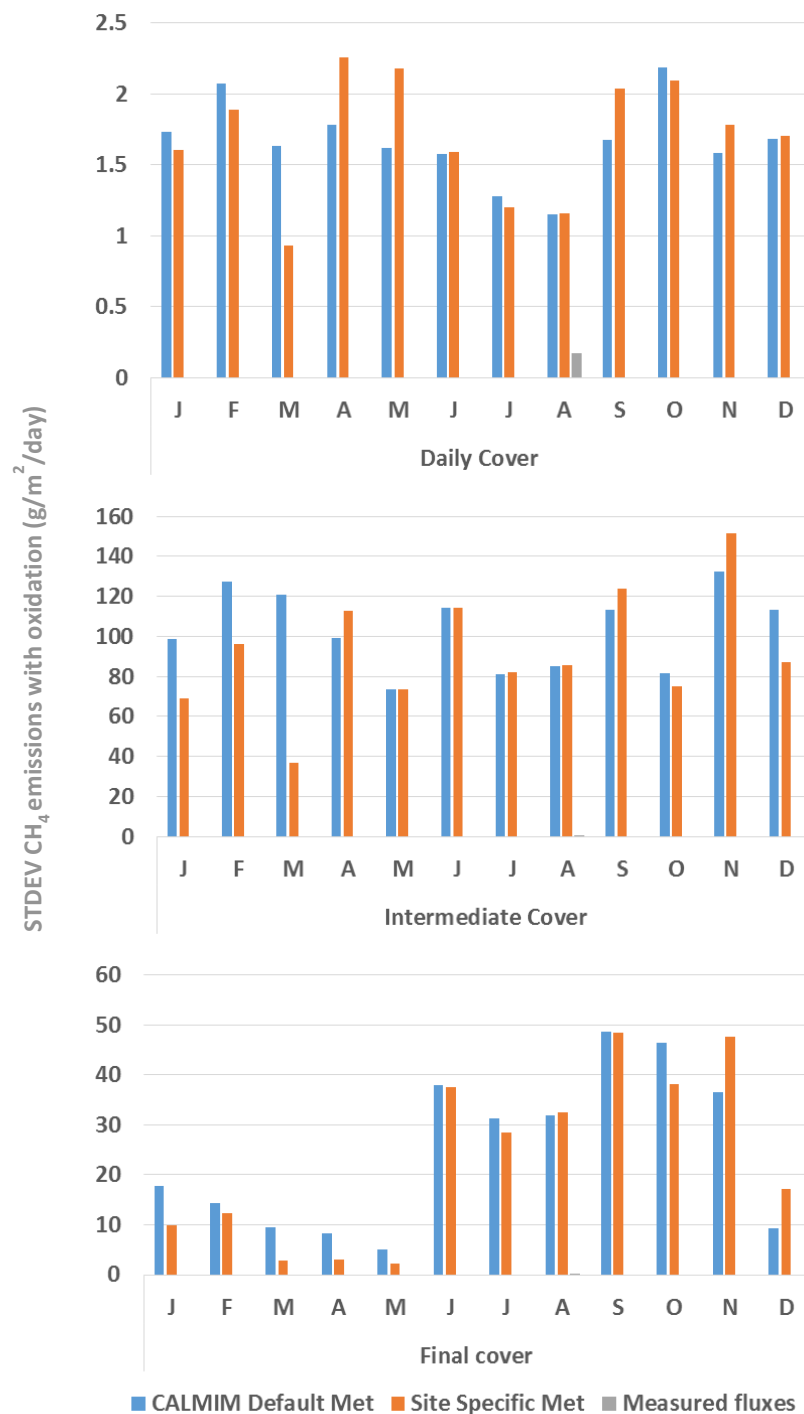


Figure 3.13: Modelled flux monthly emissions using CALMIM compared to measured flux emissions at the Marie Louise landfill site. Measured emissions are comparable to emissions associated with daily cover and are much lower for intermediate and final cover soils.

The low flux rates measured by Fourie and Morris (2004) resulted in a conclusion that waste in semi-arid South Africa degrades slowly and thus hindering gas

production. This study supports the findings by Fourie and Morris (2004), finding that CH₄ and CO₂ emission fluxes to the atmosphere are very low. The low CH₄ and CO₂ surface fluxes could be attributed to the slow degradation of waste at the surface of the cell. The waste degrades aerobically and reaches methanogenesis at a later stage or once another layer of waste is disposed above. These results highlight the risk in using theoretical models in estimating emissions without the consideration of site specific characteristics such as cover material, climatic conditions and site specific CH₄ oxidation rates in cover soils. This is critical when determining the contribution of landfill emissions to national greenhouse gas inventories. As shown in this study, the current IPCC Waste Model used to determine national emissions from the waste sector over estimates the contribution of landfill emissions, sometimes by as much as 50%, when compared to actual gas extraction rates at individual landfill sites. LandGEM and GasSim over estimate between 60 % and 30% respectively. In addition, the low flux emissions to the surface show that methanogenesis occurs at a slow rate and the low fluxes could also reflect high oxidation rates of CH₄ in cover soils to CO₂. This study also shows that landfill gas management systems aid in decreasing emissions to the surface and that despite the semi-arid environment, sufficient gas is produced (albeit at a slow rate) for the implementation of LFG extraction and utilisation projects.

Chapter 4

Air quality assessment and the pollution potential of landfill sites

Air dispersion modelling of hazardous and odorous pollutants was conducted for the Robinson Deep landfill site. BTEX (Benzene, Toluene, Ethyl-benzene and Xylene), H₂S, Particulate Matter and Total Suspended Particles were modelled using Atmospheric Dispersion Modelling System (ADMS). The modelled concentrations of pollutants are compared to ambient air quality standards to determine compliance and the potential for impact on the environment. Where possible the modelled concentrations are compared to the ambient air quality monitoring that was conducted at the site from May – August 2010. The openair package in the “R” software program is used to analyse the ambient data and assess whether the Robinson Deep landfill site is a significant contributor of pollutants in the area. Results from openair are compared to dispersion modelled emissions using ADMS.

Dispersion of major pollutants from landfill site

To conduct the baseline assessment of the Robinson Deep landfill site a review of meteorological conditions and existing ambient air quality in the area was conducted. The Robinson Deep landfill site was modelled as an area source using ADMS. Receptors and buffer zones (at 200, 500 and 1000 meters) were included in the modelling to assess how far the pollution and odours extend from the site and what buffer zone is effective (see Figure 2.15 in Chapter 2).

Dispersion models require input data including meteorological data and emissions data such as source location and height, diameter and exit velocity, temperature and flow rate. Meteorology is an essential requirement and is the principal factor to the dispersion of pollutants in the atmosphere. Hourly meteorological data (including wind speed and direction; temperature, humidity, precipitation and cloud cover) from the South African Weather Service (SAWS) for the landfill site location were used for the modelling.

Meteorological overview at the landfill site

Wind roses summarize the occurrence of winds at a location, representing their strength, direction and frequency. Calm conditions are wind speeds less than 1 m/s which are represented as a percentage of the total winds in the centre circle. Each directional branch on a wind rose represents wind originating from that direction. Each directional branch is divided into segments of different colours which are representative of different wind speeds. For the current wind roses, each circle represents a 3% frequency of occurrence. Wind speed classes are represented as 0.5 – 2.1 m/s (grey), 2.1 – 3.6 m/s (yellow), 3.6 – 5.7 m/s (red) > 6 m/s (blue).

During 2011, winds originated predominantly from the north-west (13%), north-east (10%) and south-east (8%) (Figure 4.1). Wind speeds are generally moderate winds frequently exceeding 4 m/s. Calm winds occur 0.92% of the time.

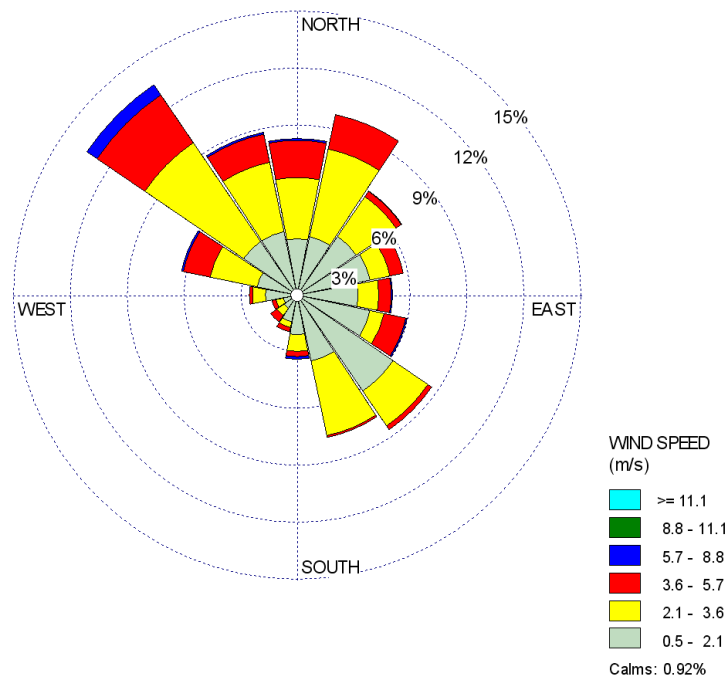


Figure 4.1: Surface wind rose from the meteorological data at Robinson Deep landfill site for 2011.

The seasonal pattern (Figure 4.2) shows winds between December and February are predominantly from the north-west (13%), north-north-east (10%) and south-east (9%). Winds are slow to moderate, with a large percentage of winds between 2 – 4 m/s. Between March and May; winds are predominantly from the north-west (15%), north-north-east (11%) and the south-east (9%). Winds are slow to moderate, with a few winds exceeding 6m/s from the north-west. Between June and August, winds are predominantly from the south-east (19% of the time), a small percentage occurs from the north-west (8% of the time). Winds are moderate; however, there is an increase in winds exceeding 6 m/s. Between September and November the predominant wind pattern is again from the north-north-east (18%)

and north-west (16%). Winds are faster with winds from the north-west exceeding 6 m/s.

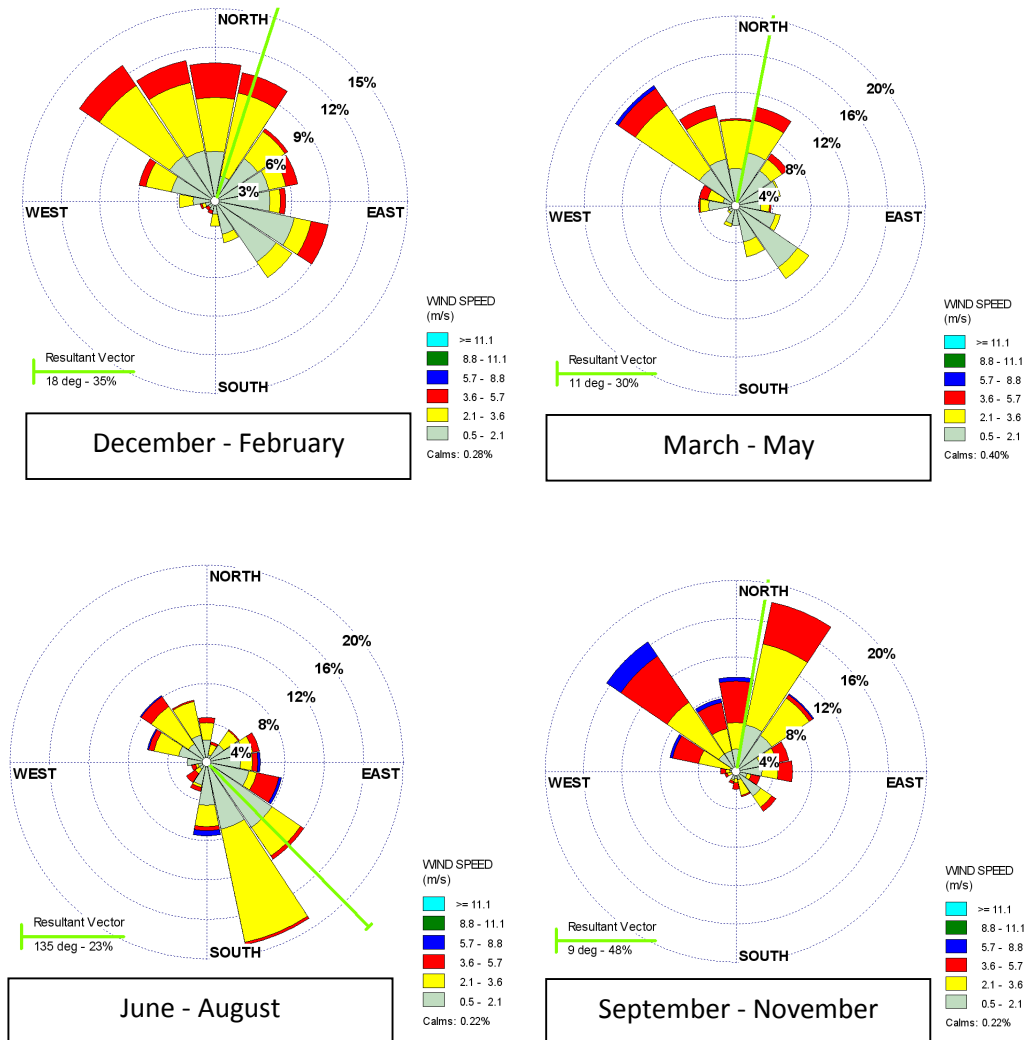


Figure 4.2: Seasonal wind roses from meteorological data at Robinson deep landfill site for 2011.

A distinct diurnal signature is evident at the site (Figure 4.3). Between 00:00 – 06:00, winds are generally calm to moderate originating predominantly from the south-east (15%) and north-east (10%). Wind speeds increase between 06:00 – 12:00 with a shift in the wind field to predominantly north-westerly (18%). An increase in wind speed is observed during this period with winds frequently

exceeding 6 m/s. Winds remain north-westerly (26%) in the afternoon (12:00 – 18:00). In the late evening (18:00 – 24:00) winds are predominantly from the north-east and south east, 13% and 10% respectively.

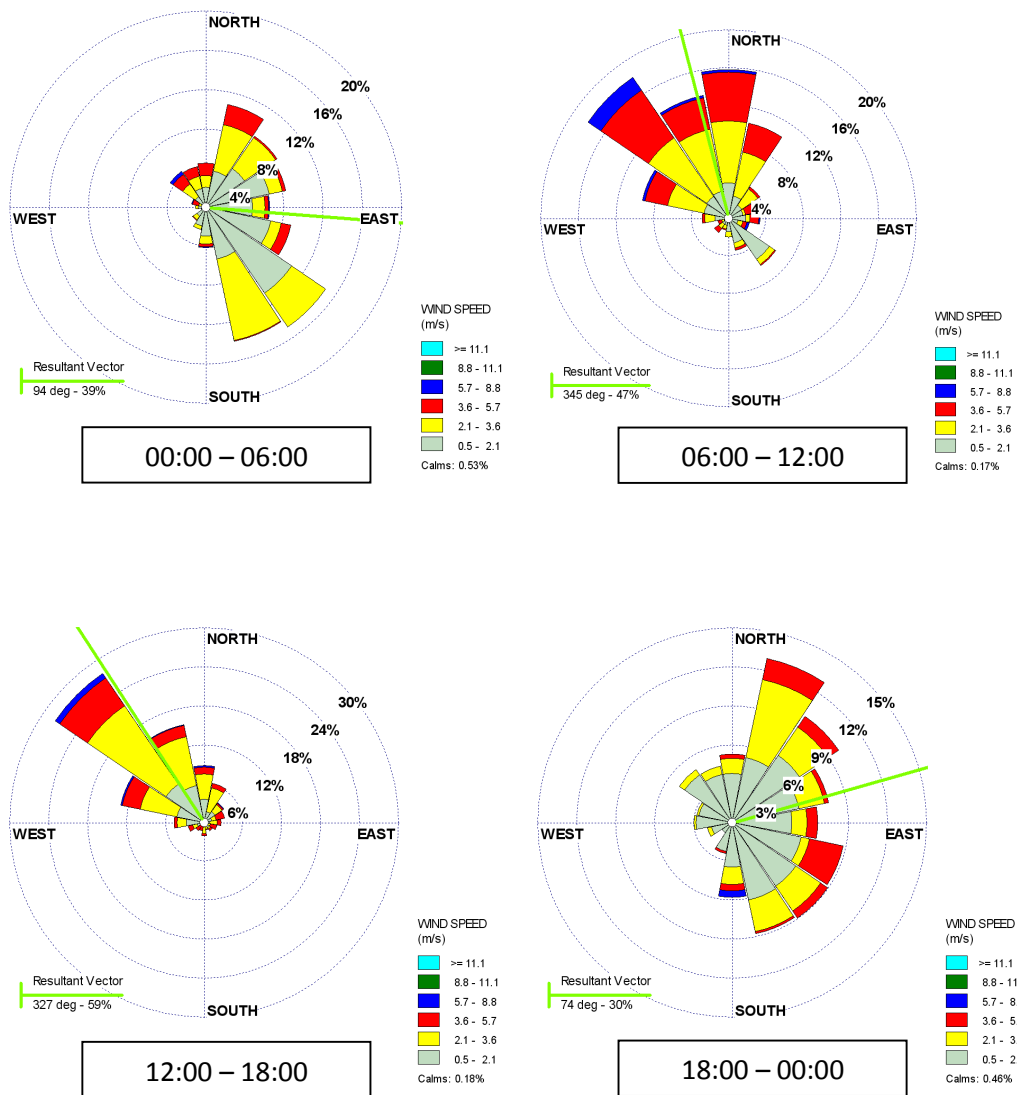


Figure 4.3: Diurnal variation of winds at the Robinson Deep landfill site for 2011, between 00:00 – 06:00 (top left), 06:00 – 12:00 (top right), 12:00 – 18:00 (bottom left) and 18:00 – 00:00 (bottom right). Winds are highly variable and increase in speed throughout the day.

Air quality standards

Ambient air quality standards highlight the maximum ambient concentrations of pollutants which are not to be exceeded (or within allowable frequencies for exceedance per pollutant) during a specified time period – usually as 24 hour or annual averages). Air quality standards and guidelines are important to effectively manage air quality in an area as it prescribes safe exposure levels for the majority of the population.

The standard ensures that negative impacts to health and the environment are prevented. Ambient air quality standards have been issued by the Department of Environmental Affairs (DEA) for several pollutants. Air quality standards for the pollutants related to this study are provided in Table 4.1.

Table 4.1: South African Air Quality Standards (SANS 1929:2011) and World Health Organisation guideline***

Pollutant	Averaging time	Concentrations	
		ppb	µg/m3
Benzene**	Annual	1.6	5.0
Hydrogen Sulphide**	24 hour	1005	150
	Annual	-	-
Particulate Matter*	24 hours	-	75.0
	Annual	-	40.0
Total Suspended Particles*	24 hours	-	300
	Annual	-	100

Both modelled and monitored pollutants are compared to the air quality standards to measure compliance and the impact on human health. The 99th percentile is used for comparison.

Dispersion of odorous emissions

Hydrogen Sulphide (H₂S) emissions and odour impact

As a national standard has not been established for H₂S, comparison is made with existing international guidelines/standards. The World Health Organisation (WHO) has recommended a daily average guideline value of 1005 ppb. Ambient H₂S concentrations recorded during the monitoring campaign are generally below the WHO guideline with a maximum concentration of 30 ppb. Hourly averages peak at 178 ppb. The diurnal signature of H₂S shows a distinct pattern. Peak concentrations are recorded in the afternoon (Figure 4.4). Concentrations also peak mid-week.

The landfill site is thought to be a major source of H₂S in the area. This is in agreement with subsurface concentrations monitored at the landfill site showing high concentrations of H₂S (Figure 4.5). Average subsurface H₂S concentrations peak during summer at 5000 ppb. High concentrations were mainly recorded along the northern and north-eastern boundary of the landfill site. However, most probes had a zero detection of H₂S.

To determine the impact from the landfill buffer zones were used at 200m, 500m and 1000m around the landfill boundary.

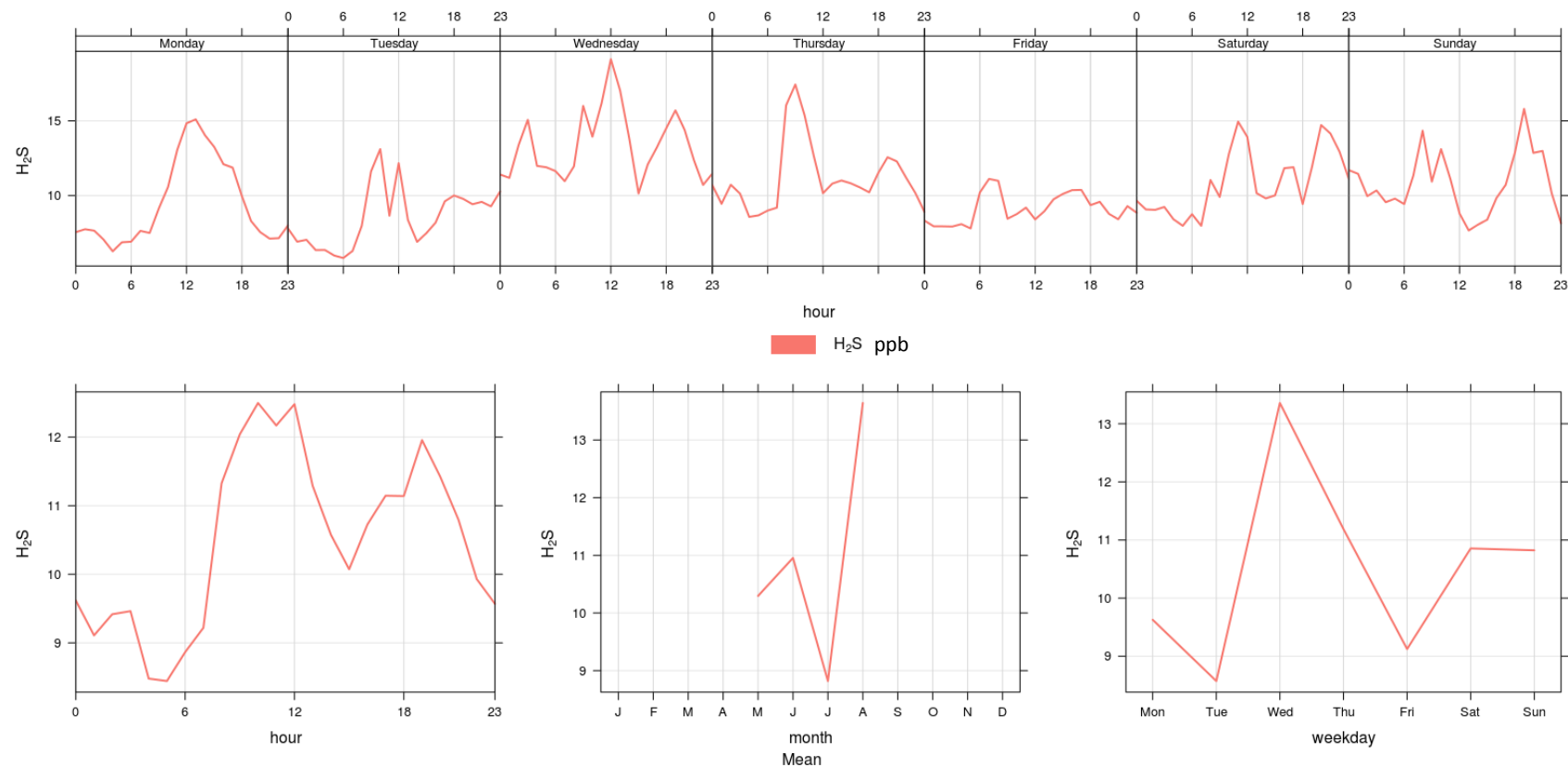


Figure 4.4: Ambient H_2S concentrations (ppb) recorded at the mobile caravan between May and August 2010. Weekly concentrations are shown on the top graph, showing mid-week peak in concentrations. Hourly averages show a diurnal cycle with peak concentrations at midday and early evening.

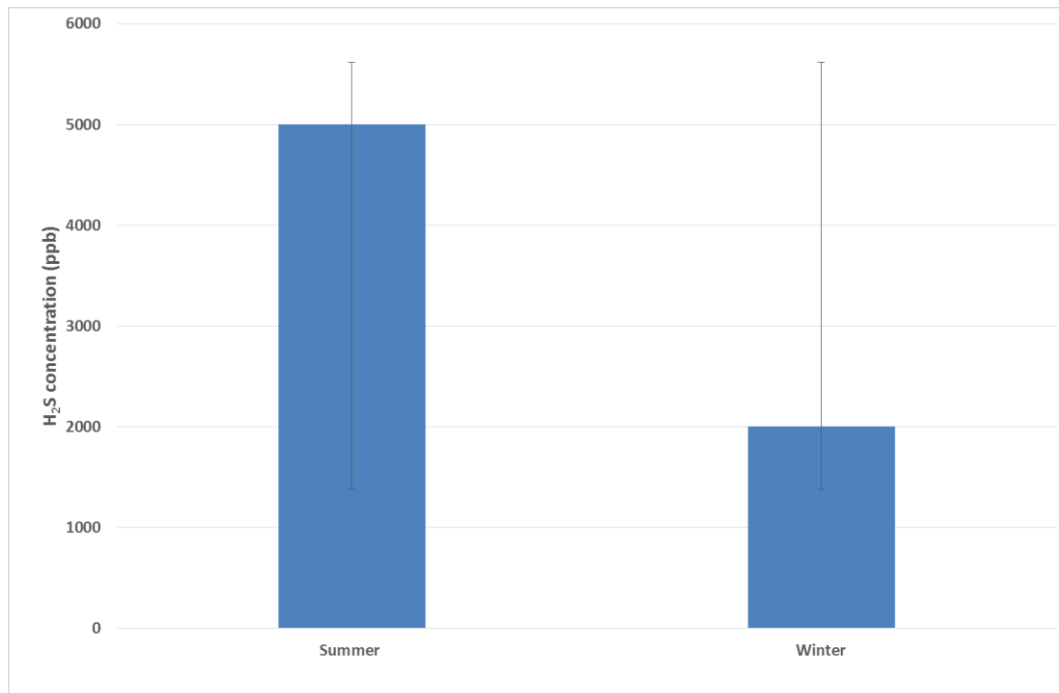


Figure 4.5: Subsurface H₂S concentrations as measured at subsurface probes at the Robinson Deep landfill sites. Error bars are given as (+/- 1 standard deviation).

Predicted pollutant concentrations for H₂S are given in Figure 4.6. H₂S concentrations are well below the 24 hour average air quality guideline. Maximum concentrations are predicted within the 200m buffer zone and do not exceed 14 ppb. H₂S concentrations were used as the primary contributor to odour at the site for the modelling. The concentrations were used to calculate odour units (OU). Table 4.2, taken from the Department of Environmental Affairs: State of Air Report (2005) highlights odour thresholds by smell recognition (using an odour panel). The most recognisable element of H₂S is its foul odour. Thresholds are divided into a 50% and 100% recognition level (DEA, 2005). However, due to this determination of an odour by the human nose the odour threshold data can vary significantly based on different methods used and the change in smell recognition from person to person (DEA, 2005).

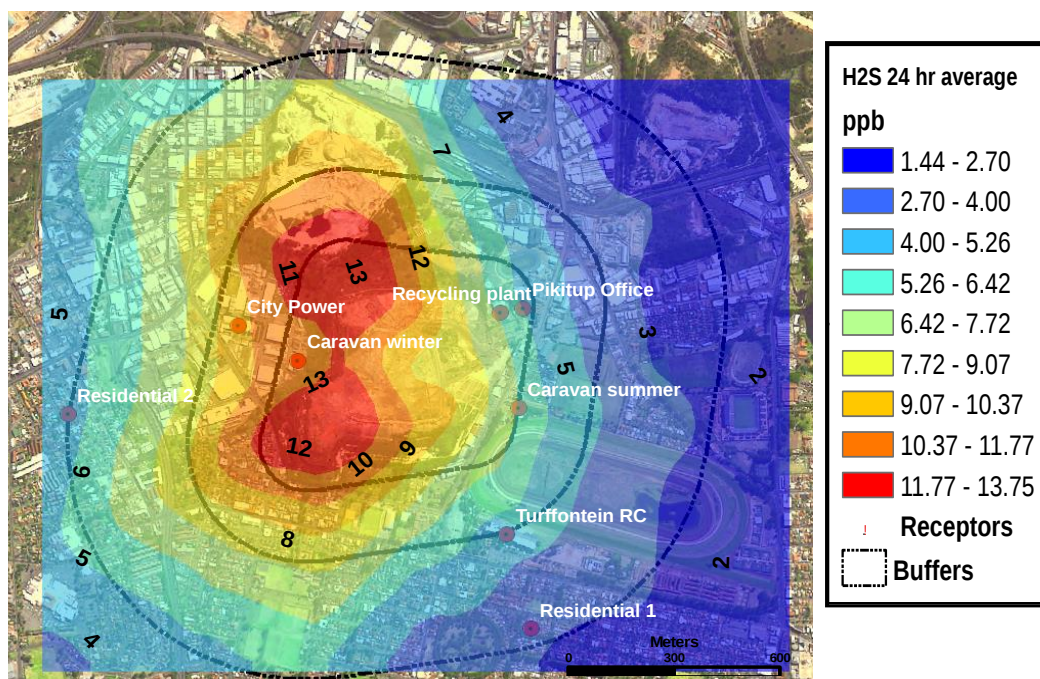


Figure 4.6: Predicted daily H₂S concentrations (ppb) at Robinson Deep Landfill. Concentrations of H₂S are fairly low and do not exceed the WHO air quality guidelines for the averaging period.

Odour modelling was considered due the fact that the air quality assessment was initiated solely based on the number of complaints of odour received by residents and businesses from the residential and industrial areas surrounding the landfill site.

Table 4.2: Odour threshold values for common sources of odour ($\mu\text{g}/\text{m}^3$)(DEA, State of Air Report 2005) used to calculate odour units for modelling (OU/m^3).

Pollutant	Odour recognition thresholds		Other odour thresholds	WHO guideline value (30 min)
	100% recognition	50% recognition		
Ammonia			500	
Hydrogen Sulphide	1 430	11.2	4.29	7

Buffer zone determination is given in Table 4.3. 5 – 10 OU/m³ is used for buffer zone determination. >20 OU/m³ are areas of significant annoyance.

Table 4.3: Odour nuisance impact determination at the Robinson Deep landfill site.

Low Impact	Medium-Low Impact	Medium-High Impact	High Impact
0-5 OU/m³	5-10 OU/m³	10-29 OU/m³	>20 OU/m³
Target range for gas management purposes	Buffer zone determination	Point of significant annoyance to a sizeable minority of the exposed population	Point of significant annoyance to a majority of the exposed population

Due to the fact that the landfill site was operational before the surrounding areas were developed, there is currently no buffer zone around the site. Industry and business complexes are situated directly adjacent to the site. The modelled dispersion shows the odour impact from the landfill to be significant (Figure 4.8). Maximum concentrations from the landfill site are predicted within the 200m buffer zone. The odour dispersion shows that a buffer zone greater than 500m is required at the landfill site towards the south-east. A number of residential areas are situated in this area and are exposed to odour nuisance from the site.

Sarkar *et al* (2003) used dispersion modelling to determine the odour impact from a landfill site in the North London and also found high concentrations greater than 20 OU/m² percentage, however these occurred at low frequencies. Roebuck *et al.* (2004) showed that odours were strongly detected between 100 and 1000 OU/m³ at the Bisasar landfill site in South Africa highlighting the significant impact of the landfill to odour nuisances in the environment.

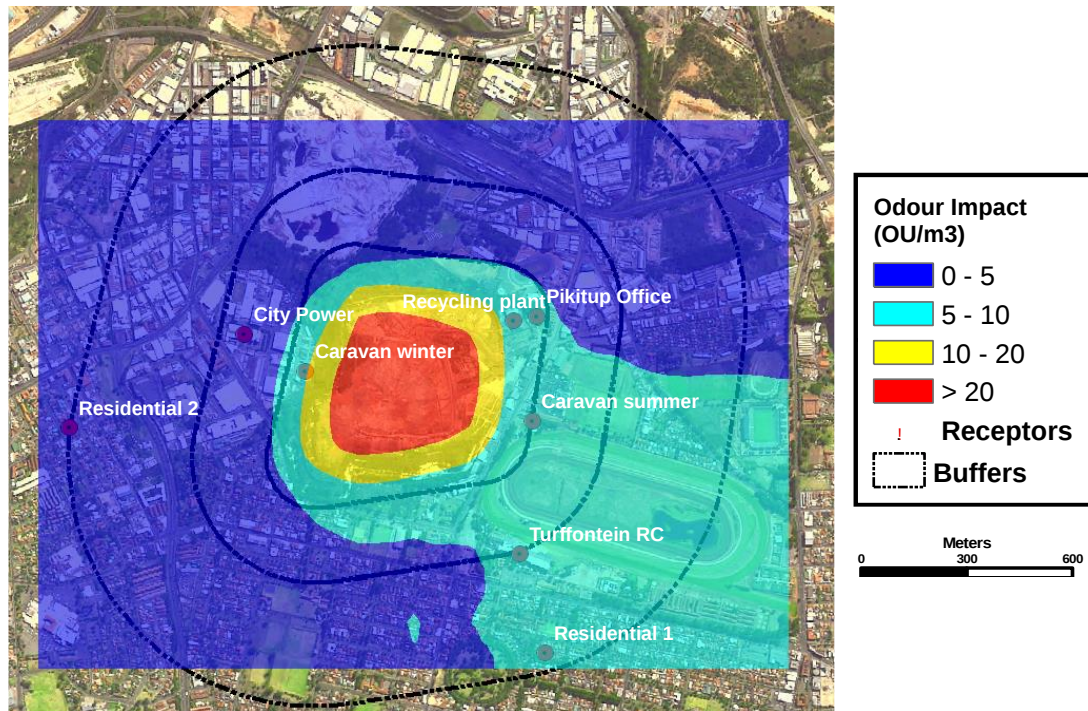


Figure 4.7: The predicted odour impact at Robinson Deep Landfill. The odour dispersion shows that a buffer zone greater than 500m is required at the landfill site towards the south east. A number of residential areas are situated in this area and are exposed to odour nuisance from the site.

To determine if the landfill site is the only source of H_2S in the area the openair package of the “R” programme, which is based on the Conditional Bivariate Probability Function (CBPF) was used. It calculates the probability that the concentration of a particular pollutant is greater than a specified value in a given wind sector including the dependence on wind speed for pollution dispersion using the bivariate polar plot (Uria-Tellaetxe and Carslaw, 2014). Each bin represents a percentile hence there are 9 bins from 10% to 100% hence values below 10% is not represented in the plots. The CBPF for H_2S shows the highest probabilities of largest concentrations (22 – 100 ppb) originate from the north-west sector with lower occurrences from the centre and south-west (6.2 – 9.6 ppb) (Figure 4.8). Hence, there is a definite additional source from the north-west contributing to odours at the site. There are a number of industries and tailings dams in the north-

east sector. Identifying the possible source in this region is beyond the scope of this work. Despite this, there is still a contribution from the landfill site in lower concentrations between 7.6 – 9.6 ppb. These are direct contributions from the landfill as the monitoring was placed at the boundary of the site. This correlates well with the predicted dispersion of H₂S in Figure 4.7 above, showing emissions between 9.07 and 13.97 ppb directly from the landfill site.

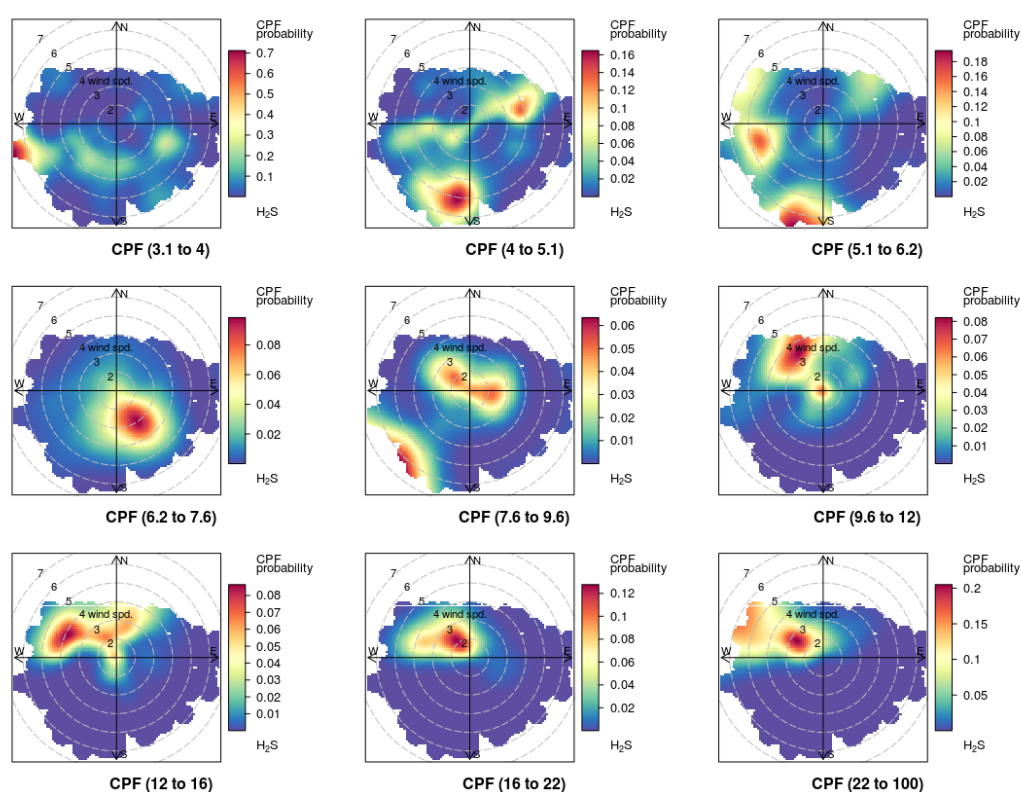


Figure 4.8: H₂S source determination using CBPF plots for different concentration intervals. Hourly averages are used from monitoring conducted in May – August 2010.

BTEX (Benzene, Toluene, Ethyl-benzene and Xylene) emissions

Predicted pollutant concentrations for Benzene, Toluene, Ethyl-benzene and Xylene (BTEX) are presented in Figures 4.9 and 4.10. The ambient air quality

guidelines set by the Department of Environment and Heritage protection in Queensland Australia are 3 ppb for benzene, 100 ppb for Toluene and up to 200 ppb for Xylene (EPA, 2008). However the World Health Organisation (WHO) recommends no safe level of benzene. Predicted benzene and toluene concentrations are low and do not exceed air quality guideline (Figure 4.9). Maximum concentrations are predicted within the 200m buffer zone. Ethyl-benzene and xylene concentrations are also low and do not exceed air quality guidelines and standards (Figure 4.10). Monitored ambient benzene concentrations recorded at the site has a monthly average of 0.56 ppb, toluene 2.75 ppb, xylene 0.58 ppb and ethyl-benzene 0.39 ppb between March and August 2010. Peak instantaneous concentrations occur during August at 60 ppb (Figure 4.11). The primary anthropogenic emissions of BTEX are from motor vehicles, aircrafts, and cigarette smoke. BTEX compounds are created and used during the processing of petroleum products and during the production of consumer goods such as paints and lacquers, thinners, rubber products, adhesives, inks, cosmetics and pharmaceutical products (Queensland EPA, Environmental Protection (Air) Policy, 2008).

The CBPF plots for benzene shows the contribution of a major source from the north-west as well and south-west sectors. Highest probabilities of peak concentrations occur in close proximity to the monitoring station under calm wind conditions. Since the monitoring station was placed on the boundary of the landfill site, these emissions are interpreted to be directly from the landfill site (Figure 4.12). Previous studies at landfill sites have found levels of BTEX emissions to be relatively low.

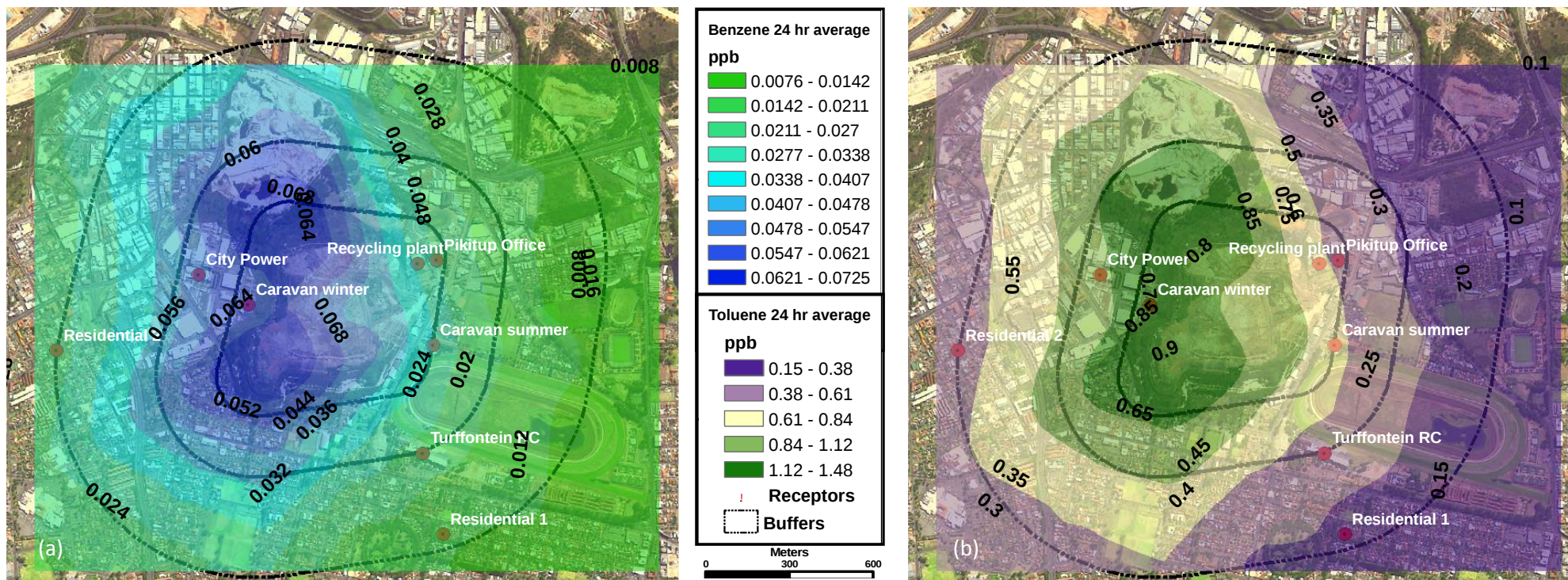


Figure 4.9: Predicted daily benzene (a) and toluene (b) concentrations at Robinson Deep Landfill. Concentrations are fairly low and do not exceed air quality guidelines for the averaging period.

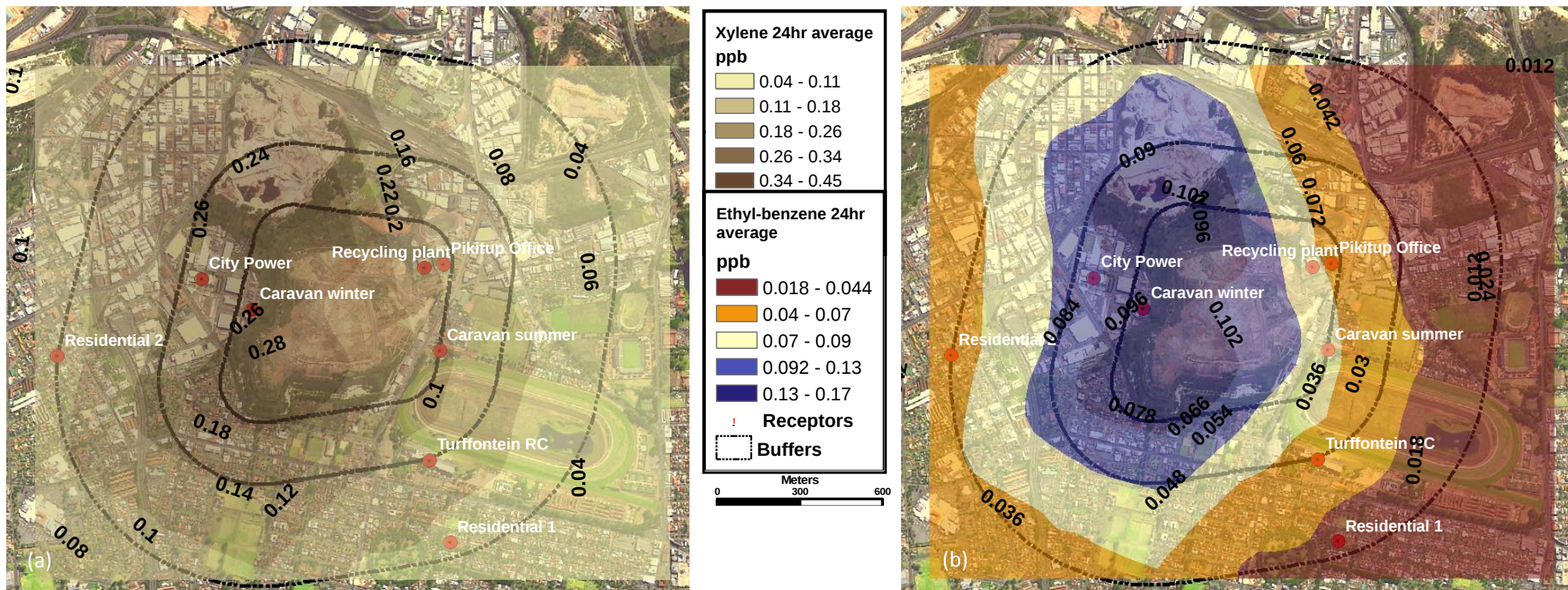


Figure 4.10: Predicted daily ethyl-benzene (a) and xylene (b) concentrations at Robinson Deep Landfill. Concentrations are fairly low and do not exceed air quality guidelines for the averaging period.

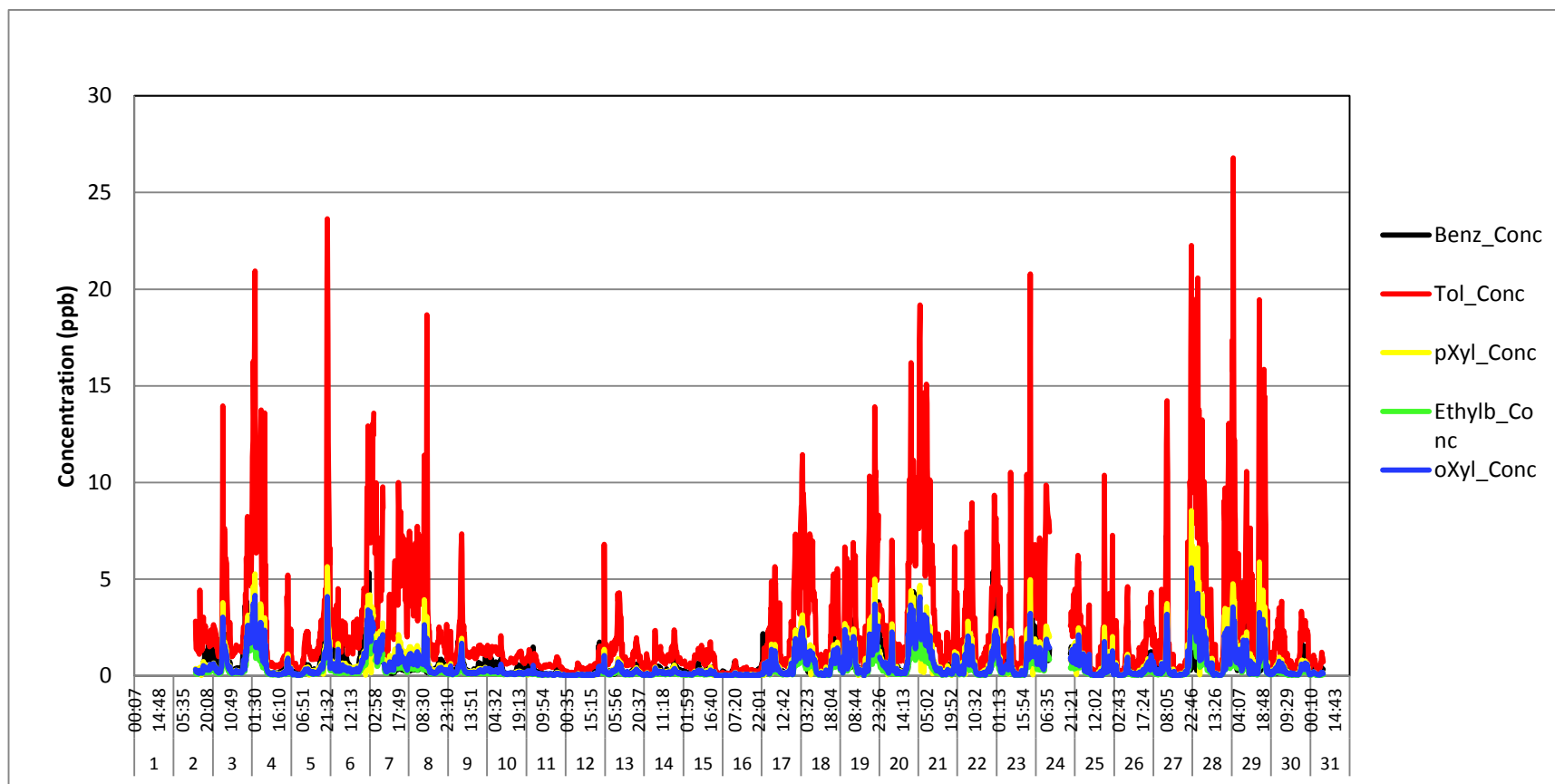


Figure 4.11: BTEX concentrations recorded at the monitoring station for July 2010.

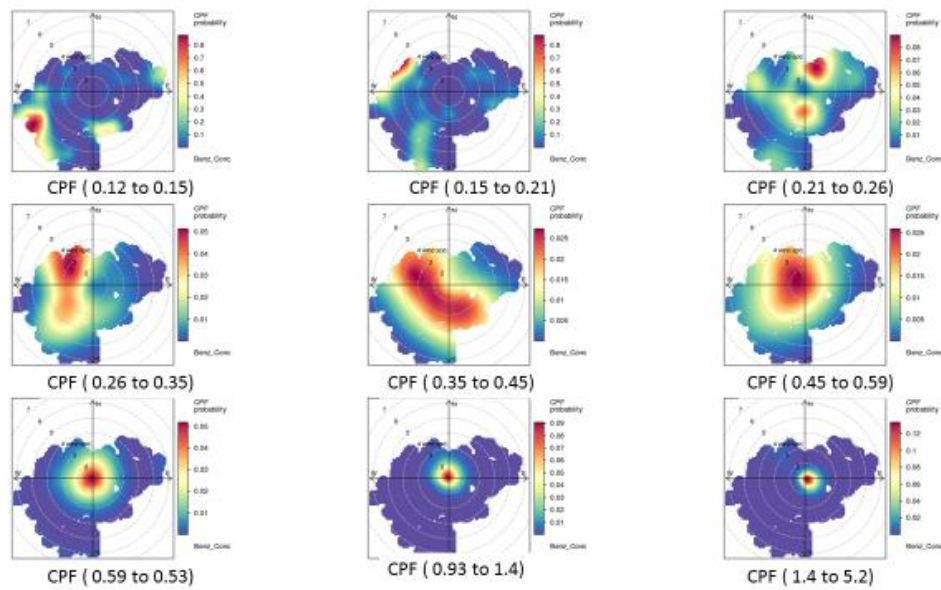


Figure 4.12: Benzene source determination using CBPF for different concentration intervals.

According to Durmusoglu *et al.*, (2009), a study conducted at a landfill site in Turkey showed that emissions of BTEX from the landfill site was not a health risk to personnel working at the site and the concentrations of BTEX measured did not pose adverse health effects to residents living nearby. Toluene CBPF shows the contribution of a major source from the north-west as well as a contribution from the landfill site. As with Benzene, the highest concentrations occur under very low wind speed conditions with a slight westerly origination (Figure 4.13). Ethylbenzene (Figure 4.14) and Xylene (Figure 4.15) show a similar trend in distribution under different wind speeds as Toluene. Highest concentrations occur in close proximity to the monitoring station under calm wind conditions with a slight westerly component. Based on this it can be seen that the landfill is not the primary source of BTEX in the area as with H₂S. The busy M1 and M2 highways are to the north- west as well as large industrial areas.

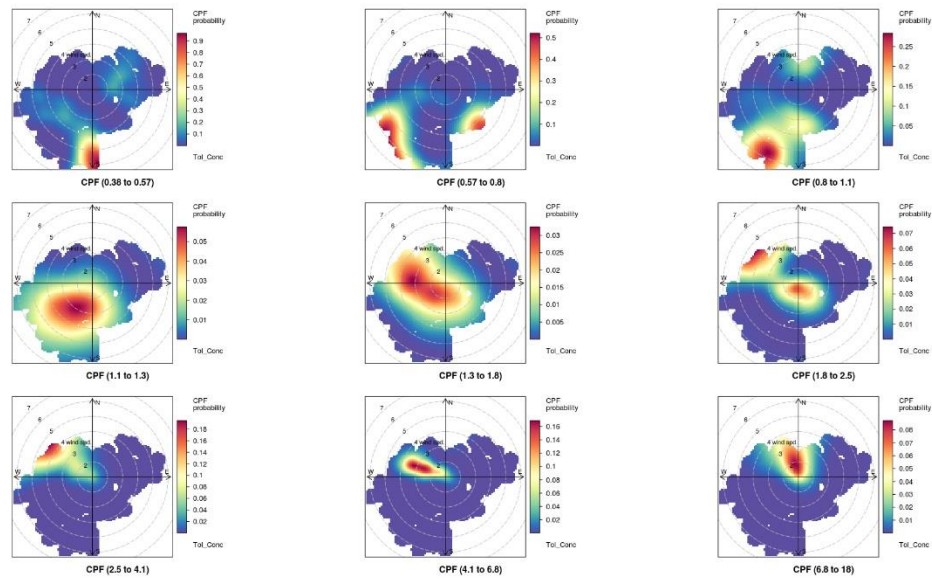


Figure 4.13: Toluene source determination using CBPF plots for different concentration intervals.

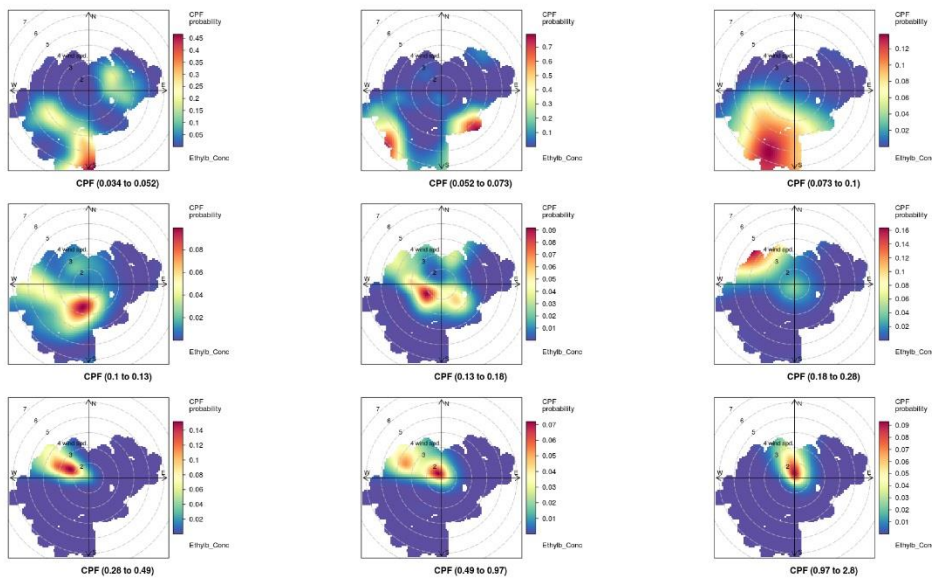


Figure 4.14: Ethyl-benzene source determination using CBPF plots for different concentration intervals

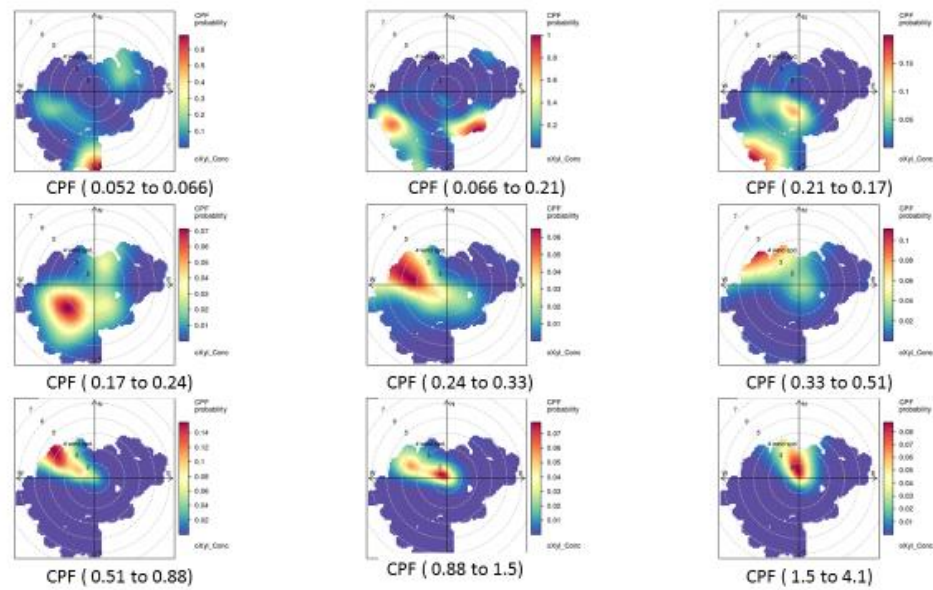


Figure 4.15: Xylene source determination using CBPF plots for different concentration intervals.

Dispersion of Particulate Matter (PM10 and TSP)

Predicted highest daily PM₁₀ concentrations range from 17.7 – 23.3 $\mu\text{g}/\text{m}^3$ whilst annual average concentrations range between 11.4 – 16.4 $\mu\text{g}/\text{m}^3$ well below the daily and annual average air quality standards of 75 $\mu\text{g}/\text{m}^3$ and 40 $\mu\text{g}/\text{m}^3$ respectively (Figure 4.16). The highest daily TSP concentrations predictions range between 39.7 $\mu\text{g}/\text{m}^3$ and 53.0 $\mu\text{g}/\text{m}^3$. Annual average concentrations range between 20.5 – 30.8 $\mu\text{g}/\text{m}^3$ (Figure 4.17). Both the daily and annual TSP concentrations do not exceed their respective air quality standards. Highest concentrations are experienced in close proximity to the site within the exclusion zone buffer of 200m.

TSP emission from the landfill site is estimated to be approximately 38 tonnes per annum. Wind-blown dust from the landfill also does not exceed the dust fall guidelines set by DEAT (Figure 4.18). However, maximum deposition rates range

between 67 mg/m²/day and 108 mg/m²/day which are associated with slight to moderate dust fall. The wind pattern, as shown in the meteorological overview, generally blows in all directions at the site with a strong north-westerly and south-easterly influence. Pollutant concentrations and deposition rates are well below both the residential and industrial limits set in SANS 1929:2011 (Table 4.4). A 200m buffer zone around the site for dust nuisance will be effective.

Table 4.4: Dust fall out rate divided in four bands according to SANS 1929:2011.

Band Number	Band Description Label	Dust-Fall Rate (D) (mg/m ² /day), 30-day average)	Comment
1	Residential	D < 600	Permissible for residential and light commercial areas
2	Industrial	600 < D < 1 200	Permissible for heavy commercial and industrial areas
3	Action	1 200 < D < 2 400	Requires investigation and remediation if two sequential months lie in this band, or if there are more than three occurrences in any 12-month period
4	Alert	2 400 < D	Immediate action and remediation required following the first exceedance, with an incident report to be submitted to the relevant authority

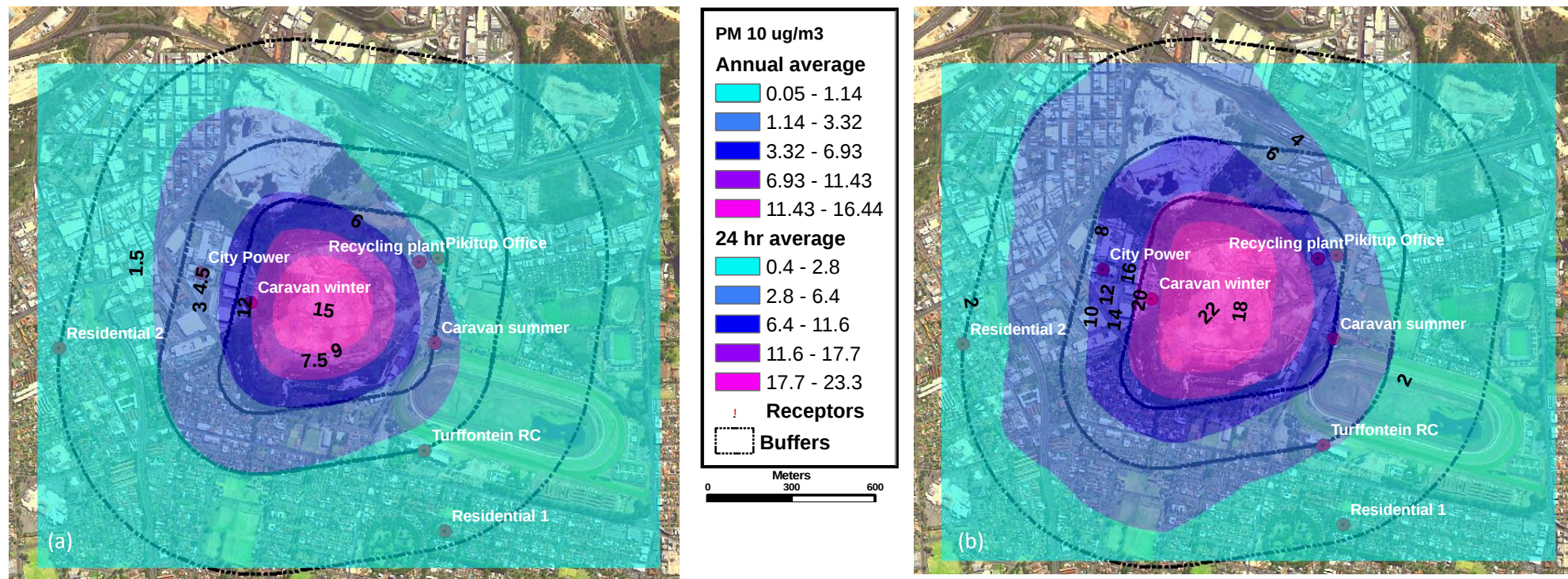


Figure 4.16: Predicted annual (a) and daily (b) PM10 concentrations ($\mu\text{g}/\text{m}^3$) from the site. Concentrations are fairly low and do not exceed the national air quality standards for the averaging period.

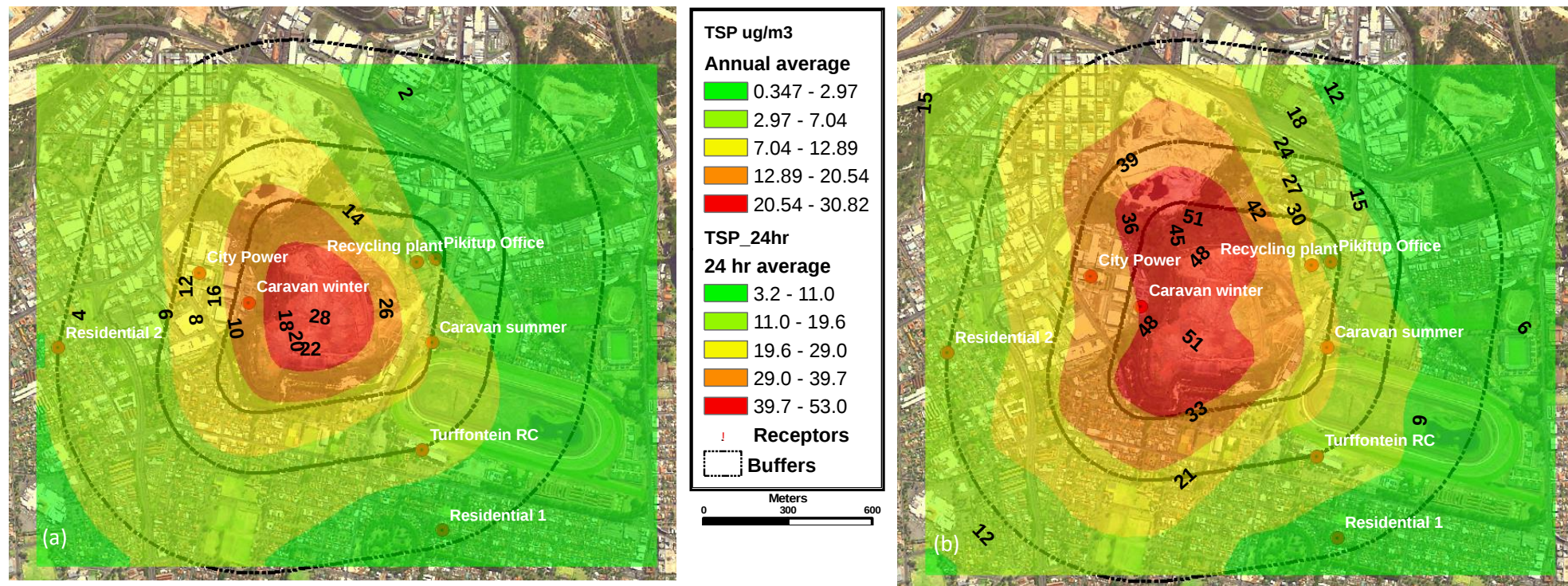


Figure 4.17: Predicted annual (a) and daily (b) TSP concentrations ($\mu\text{g}/\text{m}^3$) from the site. Concentrations are fairly low and do not exceed the national air quality standards for the averaging period

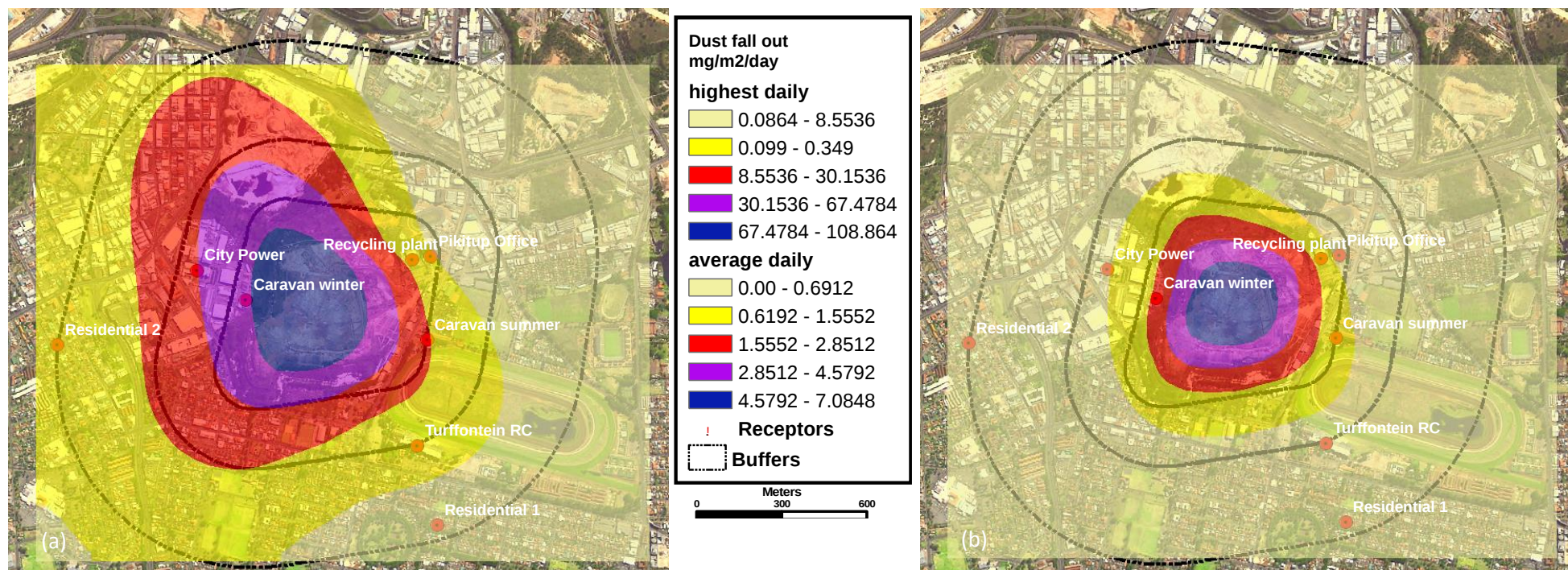


Figure 4.18: Predicted average daily (a) and highest daily (b) dust fall due to deposition of wind blown dust (mg/m²/day) from Robinson Deep. Concentrations are low and do not exceed the national air quality standards

Daily average PM_{2.5} concentrations are below the proposed daily average standard of 65 µg/m³ over the monitoring period with hourly averages peaking at 60 µg/m³. A large amount of data is missing in the data set due to the pump on the instrument failing. The pump was replaced and monitoring continued. Daily average concentrations peak at 70 µg/m³. A distinct diurnal signature in PM_{2.5} concentrations is recorded at the monitoring site. Elevated concentrations are recorded during the late evening and early morning between 22:00 and 02:00 hours then decreasing to a peak concentration at 08:00 hours in the morning. Concentrations decrease in the afternoon periods followed by a peak in the evening (Figure 4.19).

In terms of health effects, particulate air pollution is associated with complaints of the respiratory system (WHO, 2000). Particle size is important for health because it controls where in the respiratory system a given particle deposits. Fine particles are thought to be more damaging to human health than coarse particles as larger particles are less respirable in that they do not penetrate deep into the lungs compared to smaller particles (Manahan, 1991). Larger particles are deposited into the extrathoracic part of the respiratory tract while smaller particles are deposited into the smaller airways leading to the respiratory bronchioles (WHO, 2000).

PM 2.5 CBPF plots shows the contribution of a major source from the north-west and westerly direction. The highest concentrations (83 – 149 µg/m³) occur under very low wind speed conditions with a slight westerly origination (Figure 4.20). This is primarily the impact from the landfill site.

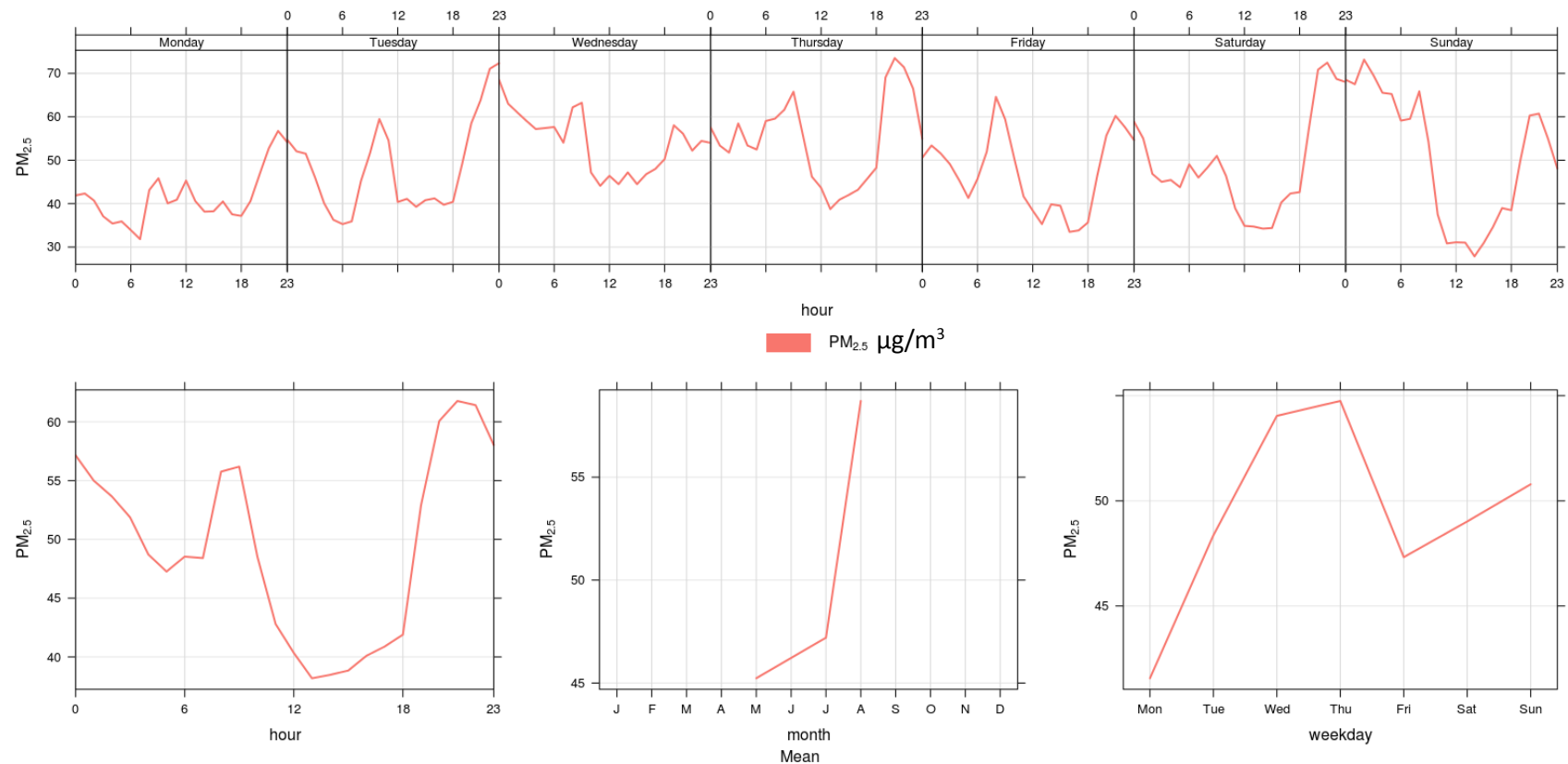


Figure 4.19: Ambient particulate matter concentrations recorded at the mobile caravan at Robinson Deep landfill site between May and August 2010. There is no particular weekly pattern to concentration, however, the diurnal pattern shows maximum concentrations mid-morning and increasing again late evening.

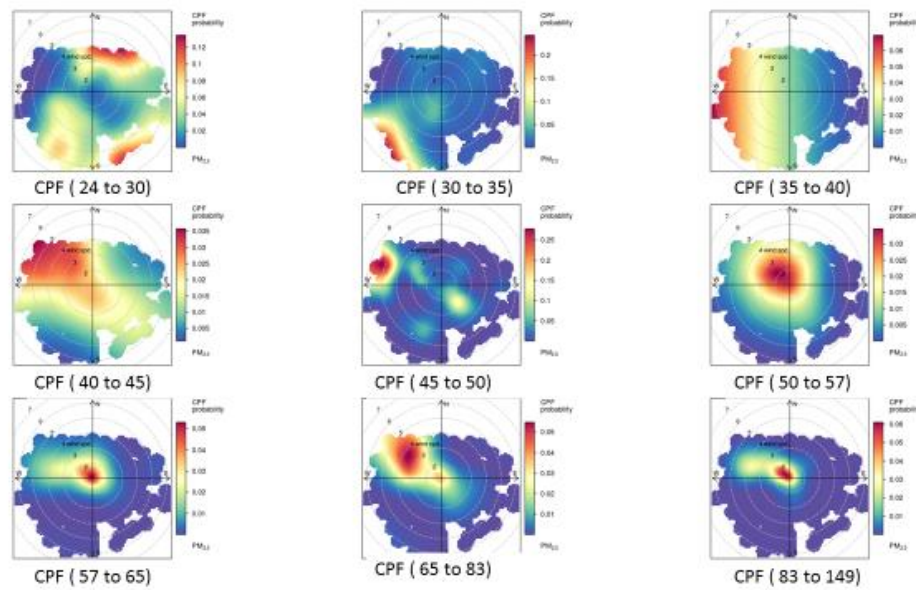


Figure 4.20: Particulate matter (PM 2.5) source determination using CBPF plots for different concentration intervals.

Based on this it can be seen that the landfill is not the only source of particulate matter although it has the highest probability to contribute to particulate matter emissions in the area. There are a number of tailings dams in the westerly and north-westerly directions which are most probably adding to the particulate measurements at the monitoring station.

Methane (CH₄) emissions

As CH₄ emissions from the landfill are analysed and discussed in great detail in Chapter 3 – only the CBPF plot is presented and discussed here as a proxy for emissions of other pollutants.

The CBPF plot for CH₄ show that highest concentrations are directly from the landfill site in correspondence with BTEX. There is also a contribution from

sources in the north-west and north-east. As with BTEX this signal source could be from the busy M2 and M1 highways that are located to the north of the landfill site. The N17 highway is also in the north-easterly direction from the monitoring station.

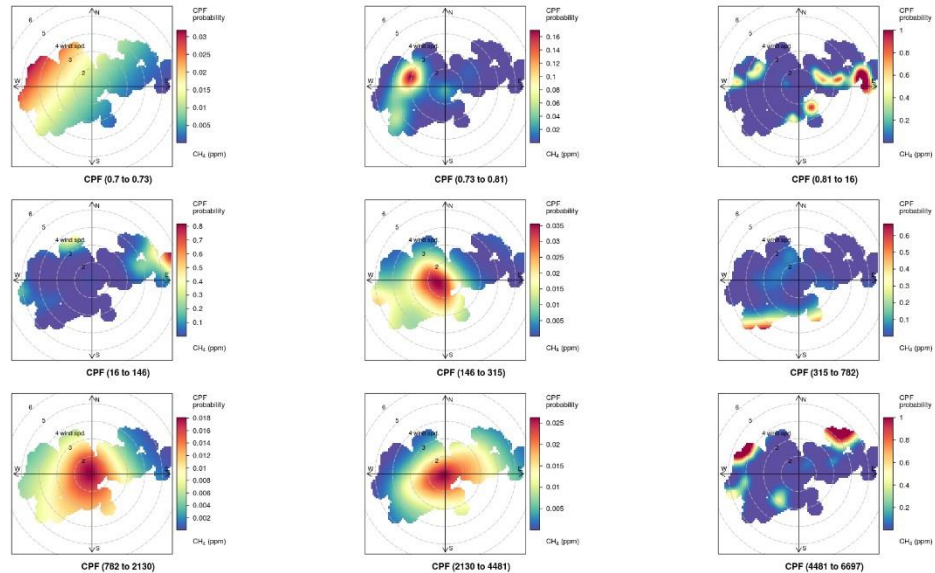


Figure 4.21: Methane (CH_4) source determination using CBPF plots for different concentration intervals.

Based on the air dispersion modelling, ambient air quality monitoring and source apportionment it can be shown that whilst the Robinson Deep landfill site is a contributor to major pollutants in its surroundings, there is a significant impact in the same area from other sources due to the prevailing wind directions and industrial activities in close proximity. Even though the predicted pollutant concentrations are generally low at the site it is important to note that these are simulated concentrations from the landfill site. Background concentrations of the pollutants were not included in the modelling and therefore predicted concentrations should be considered as an addition to background concentrations. Despite this, the source apportionment, using openair, at various concentration intervals show that the

landfill more or less contributes to the site in lower concentrations except in the case of BTEX.

The odour impact from the landfill is beyond the 500 meter buffer zone. This section of the thesis was initiated due the large number of complaints of odour from residents and people working in close proximity to the site. Whilst the landfill is a contributor to the odour issues there are also other sources north-west of the landfill. The odour complaints could also be associated with freshly disposed waste that was not covered due to various reasons. Hence, the odour pollution from the landfill is largely dependent on the manner in which the facility is managed. Dust mitigation strategies should be in place. Water tankers should be used for dust suppression at all times. To reduce gas emissions to surface especially odour nuisances, daily covering of waste with adequate cover material is necessary. Putrescible waste should be covered immediately. Odours from stored leachate can be minimised by applying odour controls to the leachate pond. In the long term, landfill gas extraction and flaring could be effective in controlling gas emissions and odours from the landfill site (McKendry *et al.*, 2002).

Chapter 5

Landfill gas extraction and application as Clean Development Mechanism projects in South Africa

Landfill gas (LFG) extraction and utilisation projects implemented in South Africa are assessed. Five projects in South Africa i.e. Ekurhuleni Metro Municipality, City of Johannesburg, eThekweni Metro Municipality, Chloorkop and Alton landfill sites are evaluated to gauge the applicability and success of each project under the Clean Development Mechanism (CDM) programme of the Kyoto Protocol. The operational expenditure and capital costs as well as the sale of emission reduction (VERs and CERs) and income or savings from electricity generation is considered. The overall Global Warming Potential reduction is also calculated.

Emission reductions from LFG extraction in South Africa

There are currently five registered (consisting of ten landfill sites) CDM LFG extraction and utilisation projects in South Africa. Two in Kwa-Zulu Natal and three in Gauteng. Emissions reductions were obtained from the municipality and/or contractors employed by the municipality to operate the LFG systems.

The Ekurhuleni Metropolitan Municipality (EMM) established a LFG extraction and utilisation project at four of its landfill sites: - Weltevreden, Simmer and Jack, Rooikraal and Rietfontein. Assessment of the potential for landfill gas (LFG) extraction and utilisation at EMM landfill sites began in 2005 when the Municipality conducted a feasibility study and trial of landfill gas extraction and

flaring. The study found that gas extraction and power generation were feasible and financially viable under the CDM, which would allow for revenue to be obtained from the sale of emission reductions. The decision was made to go forward with the development of a LFG CDM project. The construction of the full scale project started on 29 August 2007. CDM registration was finalised in February 2011 retrospectively to 26 October 2010. The CDM Project was intended to be implemented in 2 phases. In phase 1, the LFG was to be extracted and flared. In phase 2, the intention was to use the LFG for power generation and export electricity to the municipal grid. Phase 2 was commissioned in September 2014. The Project's first verification (comprised of all of the VERs generated before registration and all the CERs generated from the time of registration through 29 February 2012) has been ongoing since May 2012. A total of approximately 150,000 VERs and 60,000 CERs are expected to be verified.

The emission reductions produced by the project are given in Figure 5.1. Between October 2010 and November 2011 no emission reductions were generated at Simmer and Jack due to a sink hole developing at the flare compound. The flare was thus shut down and replaced after underpinning of the area. Since 2013 emission reductions have increased at the all sites except Rietfontein. This is due to the commission of new well installations thus adding to the gas yields at the sites. In 2012/13 eight horizontal wells were opened for gas extraction at Rooikraal. At Simmer and Jack ten horizontal wells were opened for gas extraction and four vertical wells and two horizontal wells were opened for gas extraction at Weltevreden which has contributed to the increase in emission reductions at the site.

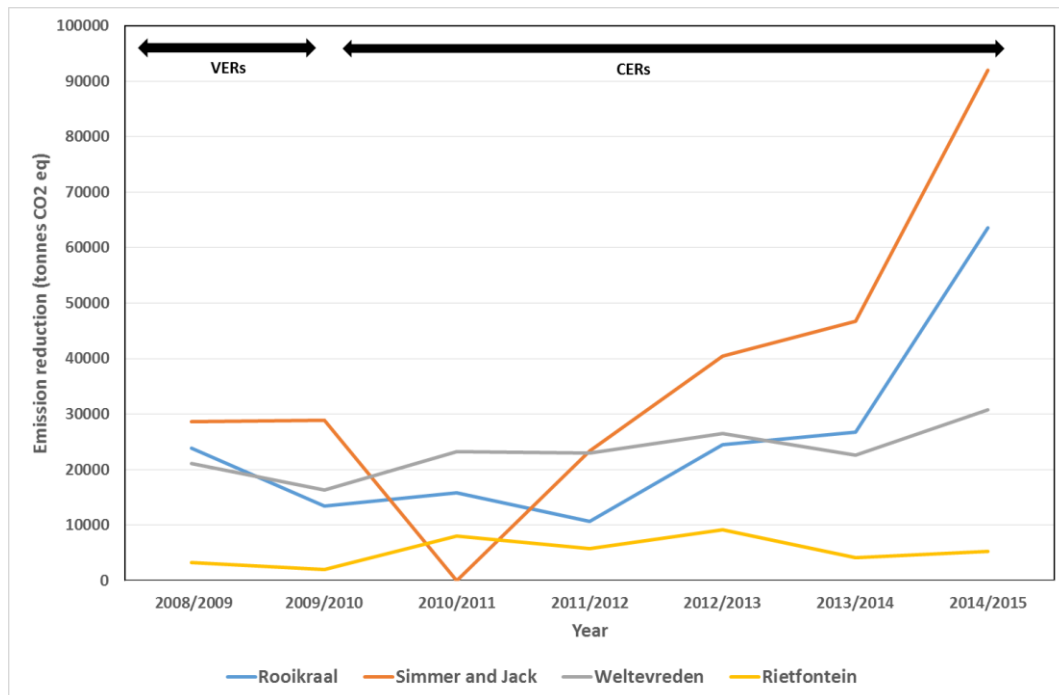


Figure 5.1: Emission reductions (for annual year July to June) produced at the four Ekurhuleni Metro Municipality landfill sites since project inception. Data presented according to municipal financial year starting July to June of the following year.

A total of 650 973 tCO₂e has been reduced by the project since start of operation up to June 2015 (Figure 5.1). Gas flow at the Rietfontein landfill site has decreased since commissioning to average gas yields approximately 200 m³/hr. The gas wells at the site are often corroded and the leachate produced block wells and condensate pumps. This could be due to the de-listed sludge waste accepted at the site. Hence, despite the LFG model predictions the waste composition has resulted in minimal gas production and making the project unsustainable at the site. LFG produced is not sufficient for electricity generation. The operational costs far exceed any possible income that could be gained from the sale of the emission reductions.

eThekweni LFG project operated and maintained by Durban Solid Waste was the first LFG extraction and utilisation project to be implemented in South Africa. The

project was implemented at three landfill sites: - Bisasar Road, Mariannhill and La Mercy landfills.

Bisasar Road is one of the biggest landfills in South Africa accepting approximately 70 000 – 100 000 tonnes of waste per month. There are 77 vertical and horizontal wells at the site and 38 leachate pumps. The project was commissioned in March 2008 and it was registered as a CDM project in March 2009. The project operates according to AM0010: “Landfill gas capture and electricity generation projects where landfill gas capture is not mandated by law” methodology. There is currently 6.5 MW engine capacity installed at the site and flow rates at 4300 m³/hr. The site will close in 2015 and should reach peak LFG production. According to Moodley *et al.*, (2015) despite the energy generating capacity being 8 MW at the site, due to the uncertainty as to how much LFG the site will continue to produce after closure (to sustain the 6.5 MW generator) no more engines will be installed. The project reduced 756 656 tCO₂e to atmosphere during the first commitment period of the Kyoto Protocol (between 2009 and 2012) and continues to reduce emissions as well as successfully generating around 45 000 MWh/year of electricity.

The Mariannhill and La Mercy landfill sites were registered as a separate CDM project. Marianhill is an active landfill site that is predicted to reach closure in 2020. The site receives between 11 000 and 18 000 tonnes of waste per month. La Mercy was closed in 2006 and has approximately 1 million tonnes of waste in place. The LFG extraction was commissioned in November 2006 and then abandoned in June 2009 due to the site not producing sufficient gas to substantiate the producing of

electricity at the site (Figure 5.2). 186 996 CERs were generated at Marianhill from project registration up to December 2013 and 7 766 CERs generated at La Mercy up to June 2009 when the project was abandoned.

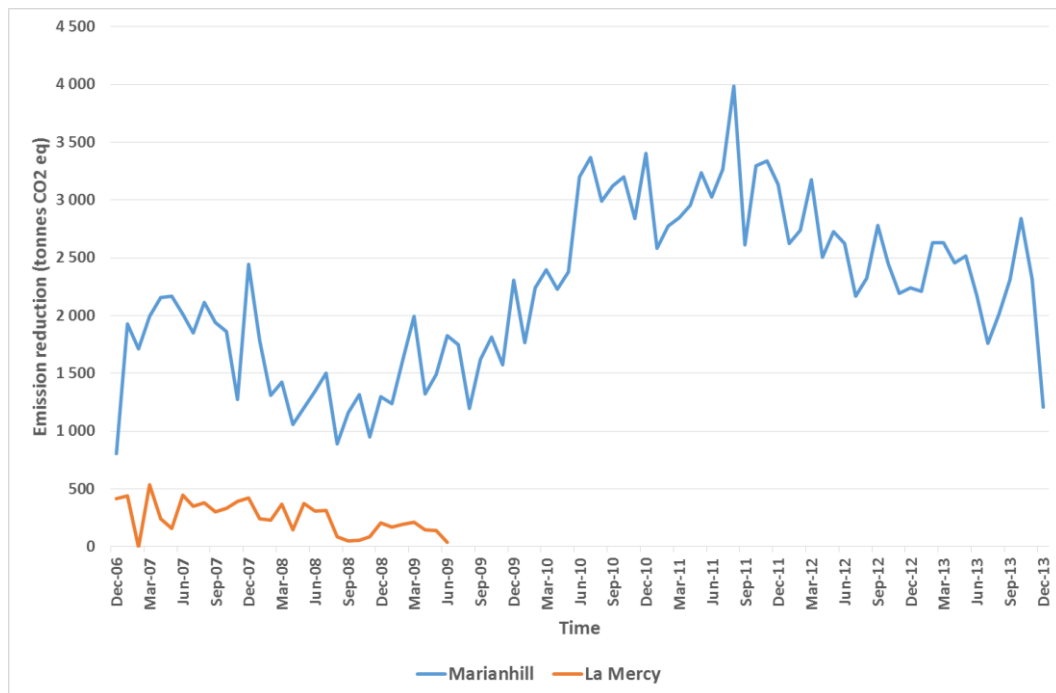


Figure 5.2: Emission reductions produced at Marianhill and La Mercy landfill sites.

A study conducted on the site in 2009 showed high levels of leachate and a significant quantity of fine Red Berea Sand used as daily cover material at the site resulted in gas extraction wells becoming blocked and as a result the engine and flare systems could not operate (Mariannhill and La Mercy Landfills PDD, 2013). Thus, only the Marianhill site continued to operate under the registered CDM project with a 1 MW engine installed at the site.

The Alton landfill site in Richards Bay in Kwa-Zulu Natal was operational from 1982 to 2004. It has approximately 1.7 million tonnes of waste in place. The project was registered as a CDM in August 2009 and commissioned in November 2009.

The gas collection systems covered the entire site as the site. The site was predicted to produce approximately 25 893 tCO₂e per annum. However, the site operated for just over two years and by February 2012 LFG extraction ceased. Power generation was successful up to the end of August 2011 and thereafter gas production was not sufficient to run the 0.4 MW engine that was installed at the site. A verification for the 3 603 tCO₂e generated while the project was operational was never completed as the volumes were not high enough to warrant a verification. According to project operators the poor performance was attributed to the fact that the site was not properly capped and would regularly turn aerobic (Cornish, 2015 pers comm). Average LFG flow rates were 100 m³/hr with CH₄ concentrations at 40% (v/v) (Figure 5.3).

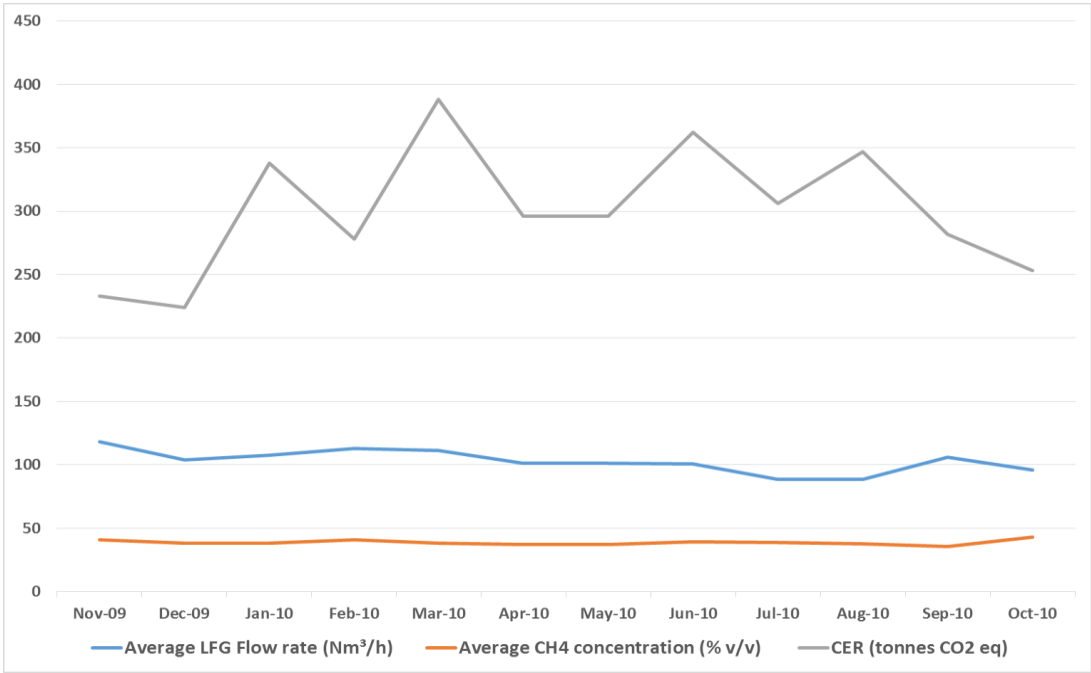


Figure 5.3: Gas production and emission reductions at the Alton landfill site in Richards Bay.

The Chloorkop landfill site is privately owned by EnviroServ and situated within EMM. The site has been operational since 1997 and receives approximately

400 000 tonnes of waste annually. The CDM project was registered on 27 April 2007. CERs for the project have been issued up to 18 January 2015. The project has no electricity generation and only flares the LFG. Vertical wells were installed in cells 1 to 3 and horizontal wells in cells 4, 5 and 6. Cell 6 is currently operational. Construction of the well field was done in a phased manner. Another 2000 m³/hr flare was commissioned in December 2008 and was fully operational in January 2009. Despite having the capacity to flare 4000 m³/hr the gas flow rate at the site averages at 1 686.8 m³/hr, and average CH₄ is 50.3% v/v (Figure 5.4). A total of 857 309.5 tCO₂e has been reduced by the project between January 2008 and July 2014 averaging at 10 852 tCO₂e per month.

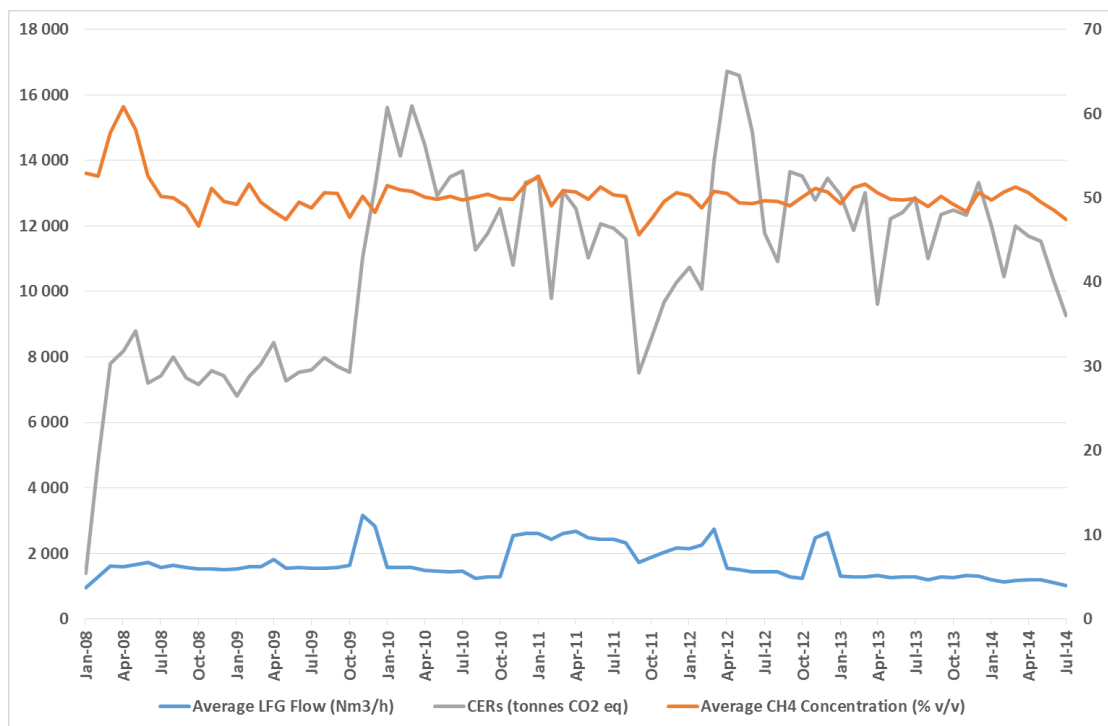


Figure 5.4: Gas production and emission reductions at the Chloorkop landfill site.

The Joburg landfill gas to energy project was awarded to EnerG Systems in 2007 which saw the private entity place all the capital and funding to purchase the

equipment for the entire gas extraction system as well as engines for future electricity generation, at no cost to the municipality. The project was registered in March 2013 retrospectively to November 2012 (before the end of the first commitment period) and consists of Robinson Deep, Marie Louise, Linbro Park, Goudkoppies, and Ennerdale landfill sites. It is estimated that all sites would reduce 542,495 tCO₂e per annum. To date only the Robinson Deep and Marie Louise landfill sites have extraction systems installed (Figure 5.5).

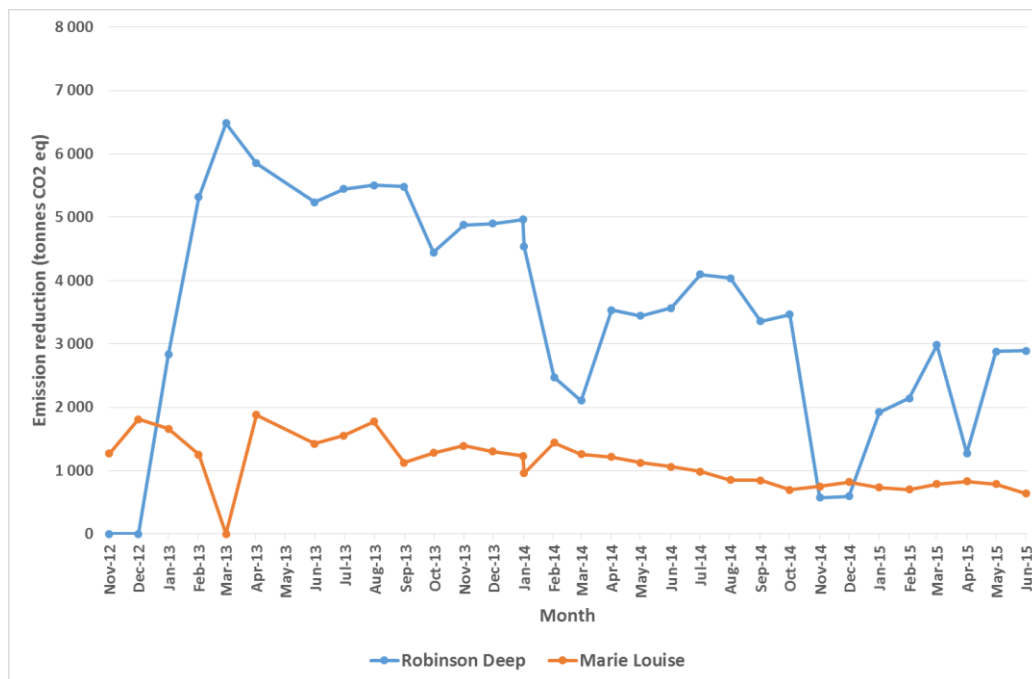


Figure 5.5: Emission reductions at the Robinson Deep and Marie Louise landfill sites in CoJ.

Gas flaring at Robinson Deep began in May 2011 with 68 horizontal and vertical wells installed linked to a 2000 m³/hr flare. Construction at Marie Louise began in February 2012 with the installation of 28 wells and the site was commissioned in April 2012. Robinson Deep performs well peaking at 6 483.4 tCO₂e in March 2013

(Figure 5.5). In total, to date, the project has generated 205 084 tCO₂e of both VERs and CERs and have sold the bulk of the emission reductions produced.

Of the ten landfill sites in South Africa that have implemented LFG extraction and utilisation projects only eight are still operational and two are successfully generating electricity. eThekweni has the most significant reduction of all projects followed by Chloorkop (Figure 5.6). However, Chloorkop is a single site whereas the eThekweni is represented by two landfill sites.

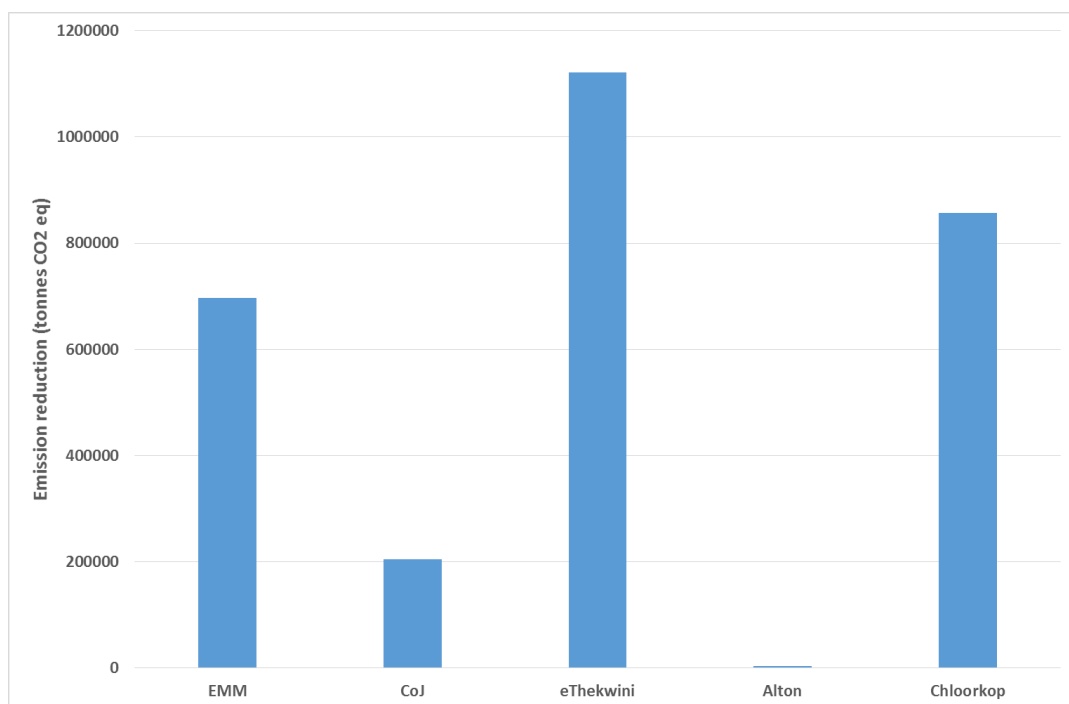


Figure 5.6: Emission reductions produced by six registered CDM landfill gas extraction and utilisation projects in South Africa. These are calculated based on emission reductions since project inception (for individual projects) up to June 2015.

There are a number of reasons for the success achieved by some projects under CDM as well as for those that have failed. Some have failed due to technical faults, failure of equipment as a result of conditions at the landfill site or due to insufficient

gas production – highlighting the fact that LFG production is very much affected by the engineering and daily operations at the sites. Inadequate compaction of waste and cover can allow air to ingress and quickly turn the site aerobic as in the case of Alton landfill site. The disposal of sludge and other leachate producing waste can cause gas wells to become blocked and corrode thus decreasing gas yields to below functional levels, as in the case of La Mercy and Rietfontein landfill sites.

Costs associated with implementation of LFG CDM projects

Most of the CDM projects were initially thought to be financially viable due to the potential sale in emission reductions on the carbon market. As the execution of LFG extraction and utilisation projects was the first of its kind in South Africa many projects initially relied on international consultants and contractors for commissioning. These skills were passed on to local companies who then continued to operate the projects on behalf of municipalities which added more costs to the project. Capital costs include the appointment of contractors for gas well installations as well as design engineers to oversee project execution. It also includes the costs associated with the procurement of all equipment including gas flares stations, gas turbine engines and gas wells.

The operation and maintenance which is overseen on a daily basis by private contractors is part of operational costs. The appointment of a DOE and the cost associated with the verification is also included in operational budget. Projects had to also appoint legal expertise for the sale of the emission reductions and negotiation of an Emissions Reduction Purchase Agreement (ERPA) and Power Purchase

Agreements (PPA) for some projects. Only two projects are municipal owned but private contractors are used for all maintenance as well as gas well installations. The eThekweni Municipality has spent over R110 million on capital costs and R8 million per annum on operational and maintenance of the systems (the project is funded by the Department of Trade and Industry Critical Infrastructure Programme (CPI), the Department of Minerals and Energy and loan funding from the French Development Bank). The full breakdown of expenditure by EMM is given in Table 5.1 and fully funded by the municipality itself. The project capital and operational expenditure is approximately R110 million since project inception.

Table 5.1: CAPEX and OPEX expenditure for EMM CDM project at four landfill sites.

Year	Capital budget	Description	Operational budget
2007/2008	28,540,084	Well field installation	
2008/2009	8,131,387	Flare installation	3,608,549
2009/2010	0		3,304,309
2010/2011	8,000,000	Phase 1 well field extension	3,444,640
Initial implementation	44,671,471		10,357,498
2011/2012	4,500,000	Phase 2 well field extension	5,000,000
2012/2013	8,500,000	Phase 3 well field extension	6,000,000
2013/2014	10,000,000	Phase 4 well field extension	6,000,000
2014/2015	2,000,000	Phase 5 well field extension	6,000,000
2015/2016	2,000,000	Well field extension at Welt	6,000,000
Phase 2-6	27,000,000		28,000,000
Total Cost up to 2015			110,028,969

This excludes the expenditure on a transactional advisor for the negotiations of the original ERPA with Endesa and for a new sale of the emission reductions which EMM are currently undergoing. It also does not include the cost of the appointment of a DOE for the verification. Whilst eThekweni has gained an income from the sale of emission reductions and electricity produced by the project (Figure 5.7), EMM has had no income since project inception.

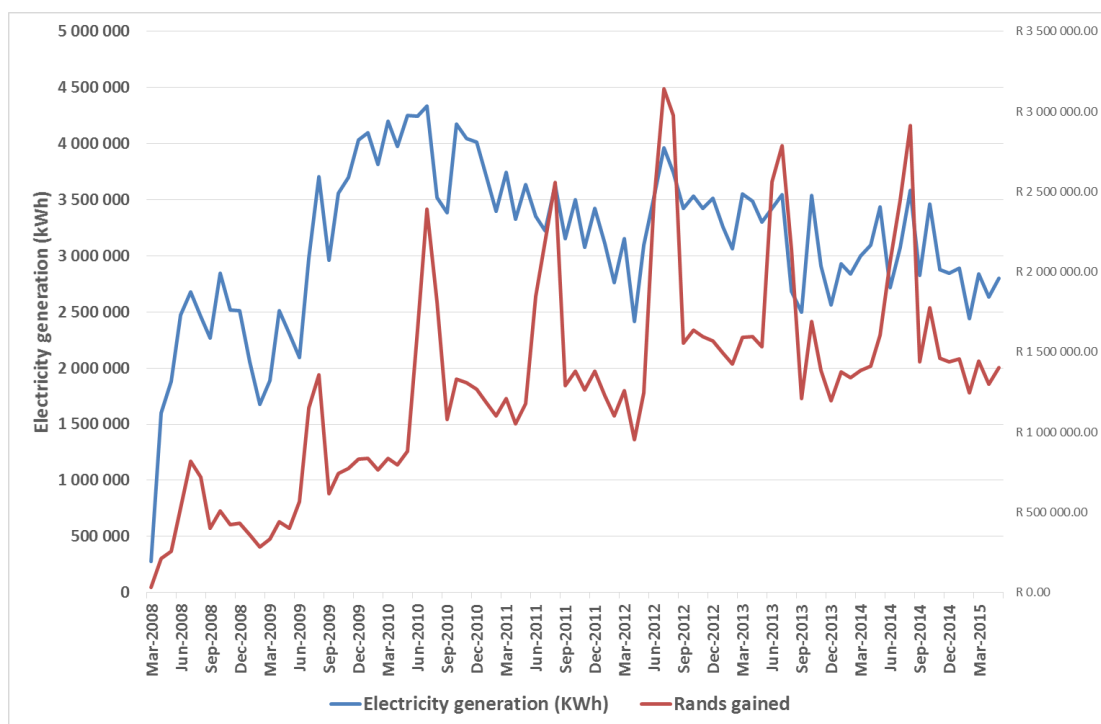


Figure 5.7: Electricity generated and income gained in Rands at Bisasar Road landfill site.

It is still undergoing the first verification which was delayed due to the procedures in appointing a DOE and since then has taken over three years to conclude. However, the project began generating electricity in September 2014 and has generated 3 698 MWh up to August 2015. At R1 a kW that is a saving of R3 698 000 to the municipality in electricity consumption.

Challenges experienced implementing LFG extraction projects

Technical challenges such as low gas production and flow rates, high leachate collection leading to blockages of wells and landfill sites turning aerobic due to improper operations during waste disposal are but a few, resulting in the termination of some projects. Effective landfill management and correct daily operational procedures (good compaction and daily covering of waste) at the landfill site is crucial for the successful implementation of LFG extraction projects. In addition, operating and maintaining an LFG project is expensive and many spare parts are obtained from overseas.

Administrative issues have also delayed projects and could also result in projects being cancelled. The CoJ for example has delayed generating electricity at the two operational projects due to a long delay in signing a Power Purchase Agreement (PPA) with Eskom and wheeling agreement with City Power. The project was successfully awarded a tender as an Independent Power Producer (IPP) for sale of the electricity at a rate that makes the project financially viable. In addition, the project registration with the UNFCCC took almost 3 years.

Registration of the EMM CDM project was also severely delayed due to several changes in methodologies and report templates by the UNFCCC as well as changes in DOE appointments (due to the cancellation of DOE's accreditation with the UNFCCC). Registration is lengthy, tedious and costly. There were further delays in appointing DOE's due to the Municipal Finance Management Act (MFMA No. 56, 2003) and Supply Change Management Policies (SCM). As most DOE's were

based internationally at that time deviations from the MFMA and SCM had to be considered. A tender process had to be conducted for the emission reduction sales which was also lengthy and time consuming.

EMM entered into an agreement in May 2007 to sell the emission reductions from the project to the Spanish electricity utility Endesa Generación (Endesa). The ERPA required EMM to deliver to Endesa at least 800 000 CERs from 2008 to December 2012 (the end of the first commitment period of the Kyoto Protocol). Endesa intended to use the CERs for compliance with its obligations under the EU Emissions Trading Scheme (EU ETS) during the first commitment period. Due to the delayed registration of the project EMM was unable to deliver any CERs to Endesa. In February 2012 Endesa served EMM with a Default Notice for Delivery Failure of CER's as per the ERPA and the ERPA was cancelled in June 2012. The predicted CERs were also never achieved and as a result the EMM adopted a turnaround strategy to increase the gas production at the landfill sites.

eThekwini CDM project was also delayed during assessment due to community opposition and at Bisasar Road. Communication and language barriers during the verification was another challenge faced by the project. The verification process is very intensive and costly. It requires a large amount of background and historical information to be supplied, including the formulas and calculations used in monthly reporting, instrument history of calibrations and repairs or replacements of equipment, data recovery, analysis and interpretation have to be verified. EMM's first verification has taken over three years and is still being finalised.

The biggest challenge faced by projects however is the downturn in the carbon market caused primarily by the global economic crisis. Carbon credit prices have plummeted and there is currently an oversupply of CERs in the EU market. Most, if not all, projects were deemed financially viable due to the sale of credits. eThekweni had sales agreements of €15/ton, EMM €10/ton and CoJ €11/ton. However, the market price for CERs is only a few euro cents per tonne of CO₂e, making most projects financially unsustainable. In addition, the cost of verification outweigh the financial profit of the CERs. Hence, currently operating and maintaining the LFG extraction projects is extremely costly and burdensome to municipalities. However, there are other benefits the projects provide and as a result they continue to operate.

Benefits of project implementation

Due to the fact that most landfill sites in South Africa are situated in and around residential areas, the implementation of the CDM project has resulted in a number of environmental and social benefits to the surrounding communities. The local air quality has improved as a result due to the extraction and combustion of landfill gas. The potential for migration of landfill gas has also been reduced thus significantly reducing the risk of explosion, toxicity and asphyxiation due to LFG accumulation. As shown in this study that both LFG migration as well as direct emissions to the surface are reduced once an extraction system is operational at the site.

The CDM projects have pioneered a new industry in South Africa. The implementation of these projects has resulted in the importation of new technology to the country and the transfer of skills to everyone involved from the operations and maintenance personnel as well as the project design engineers. The projects have created permanent and non-permanent jobs. Companies involvement have gained all round skills development and training in terms of project management, technical skills (electrical, mechanical and civils), contracting and construction management of mainly previously disadvantaged individuals.

The projects have also resulted in the indirect employment and transfer of skills by engaging with local manufacturers and suppliers in respect of the manufacture of specialised landfill gas equipment, thereby contributing to the local economy in terms of procurement and the transfer of international technologies. Many of these engineering companies have now become experts in the field of landfill gas well field installation – a skill that was previously not available in South Africa.

The flare manufacturers have also provided on-site training to relevant project personnel upon commissioning of the gas extraction, flaring and CDM monitoring systems, and also have continued with skills development and training during their service visits, resulting in continuous skills development, training and technology transfer to local personnel.

In addition, a revenue to the municipality and private owners is made especially with the production of electricity. eThekweni for example generates an income of approximately R850 000 per month by adding 6.5 MW of power to the municipal

grid every hour (Butt, 2011). Most projects have also gained an income from the sale of emission reductions, despite not gaining as much as anticipated due to the economic downturn in the carbon market.

The most beneficial aspect of implementing these projects is the large effect it has on reducing and mitigating greenhouse gas emissions in South Africa. Combined the six LFG extraction and utilisation projects have mitigated approximately 3 million tCO₂e to the atmosphere and continue to reduce emissions on a daily basis as well as providing an alternate energy source by producing electricity. According to the USEPA GHG equivalencies calculator (<http://www.epa.gov/cleanenergy/energy-resources/calculator.html>) the reduction of emissions by the landfill gas CDM projects in South Africa is equivalent to annual greenhouse gas emissions from 572 959 passenger vehicles, CO₂ from the installation of 749 wind turbines and electricity use by 374 354 households in one year. Hence, the projects make a significant contribution to reduction of greenhouse gas emissions from landfills.

Chapter 6

Summary and conclusions

Sources of CH₄ include biogenic emissions from wetlands, fossil fuel mining and burning, biomass burning, rice production, enteric fermentation from livestock and waste management (Bingemer and Crutzen, 1987; Gourlay, 1992; Neue *et al.*, 1994; Bogner *et al.*, 1995; Bogner and Matthews, 2003; Neef *et al.*, 2010; Ciais *et al.*, 2013). Anthropogenic activities contribute approximately 48.8% of the total amount of CH₄ emitted each year and landfills contribute less than 5% of global greenhouse gas emissions (Hein *et al.*, 1997; Houweling *et al.*, 1999, Denman *et al.*, 2007, Bogner *et al.*, 2007).

The uncertainties in estimating CH₄ emissions from waste disposal using theoretical models are substantial due to the input data required. Waste composition data are often not available and data on waste quantity disposed of at landfills are inaccurate and mostly unavailable. In South Africa it was found that data is often not available before 2000. However, many municipalities are upgrading and investing in proper data collection as it has become a legislative requirement and hence future emissions calculations will be based on more dependable data.

The intention of this research was to quantify LFG emissions and production from landfills in South Africa using LFG estimation models LandGEM, GasSim, the

IPCC Waste Model 2006 and CALMIM. Site specific data, where available, are used in the models to estimate LFG production. Modelled LFG production is compared to actual gas yields to assess the applicability of LFG models to South African landfill sites. Surface LFG emissions at two landfill sites were measured using flux chambers. The air quality impact from a landfill site was measured by ambient monitoring using a mobile monitoring station. The success of LFG extraction and utilisation projects under CDM is also determined.

Landfill gas emission estimates in South Africa

The IPCC Waste Model was used to estimate emissions from waste in South Africa from all provinces and the eight major Metro Municipalities.

1. 531 735 tCH₄/annum and 635 207 tCH₄/annum emissions from waste disposed at landfill sites were predicted for 2015 and 2020 respectively in South Africa (11 166 431 tCO₂e and 13 293 370 tCO₂e and for 2015 and 2020 respectively). In South Africa, according to the National Greenhouse Gas inventory there was a 72.3% increase in CH₄ emissions from landfills between 2000 and 2010, from 9 019 000 tCO₂e in 2000 to 15 535 000 tCO₂e in 2010 (DEA, 2014).
2. The distribution of emissions from waste disposal highlights the large divergences and the vast income gap between rich and poor in South Africa, with highly urbanised areas receiving the majority of waste collection services and disposal facilities hence contributing the majority of emissions predicted.

3. The Gauteng province has the largest population and waste generation rates per capita within the country and hence has the highest CH₄ emissions contributing 49.37% of all CH₄ emissions from waste disposal in South Africa in 2015.
4. The Western Cape has higher CH₄ emissions than both Kwa-Zulu Natal and the Eastern Cape despite these provinces having larger populations. This is mainly due to higher urbanisation rates within the Western Cape with larger rural areas found in the other two provinces. As a result, waste collection rates in the Western Cape at 93% far exceed both Kwa-Zulu Natal and the Eastern Cape.
5. The eight Metro Municipalities contribute 5 512 080 tCO₂e and 6 369 661 tCO₂e for 2015 and 2020 respectively with 2 846 170 tCO₂e in 2015 (approximately 48%) of these emissions emanating from the three Metro Municipalities in Gauteng province. Previous calculations by Vanyaza (2014) modelled emissions from seven municipalities in South Africa and determined the total emissions from MSW to be between 5 000 000 tCO₂e in 2015 and 6 500 000 tCO₂e in 2020.
6. The highest CH₄ emissions between all Metro Municipalities is from City of Johannesburg contributing to over 60 000 tCH₄ per annum in 2015 and exceeding 70 000 tCH₄ per annum in 2020.
7. The City of Cape Town has slightly higher emissions than eThekweni which can be attributed to the higher waste collection rates in Cape Town at 94.3%

compared to 86.1% within eThekweni. Despite the subtropical climate and higher degradation rates which add to the predicted CH₄ emissions for eThekweni.

Landfill gas emission estimations for individual landfill sites

1. LandGEM, IPCC Waste Model and GasSim were used to model emissions from individual landfill sites. The predicted generation are compared to actual gas yields at each site.
2. Predicted CH₄ generation using LandGEM is higher than both IPCC and GasSim attributed to the model applying a single *k* value for all waste degradation. CH₄ generation using the IPCC Waste Model is only slightly lower than LandGEM as despite dividing the waste into fractions it has an average *k* of 0.045.
3. Current LFG yields (at current gas well coverage) are well below predicted average gas flows when compared to LandGEM, IPCC Waste Model and GasSim. However, when considering a 70% coverage of gas wells, LFG yields are above CH₄ generation predicted by GasSim for some sites.
4. Comparing predicted emissions to actual LFG yields show that LandGEM and IPCC Waste Model over estimate CH₄ generation at all sites whilst GasSim is the closest estimation to actual gas yields at the sites. Since the GasSim model takes into consideration the differing decomposition rates for waste fractions, that is, it allows for certain percentages of the waste stream

to decay at a later stage, it more accurately predicts CH₄ generation through the life of the waste disposed at the landfill sites. It also takes into consideration the infiltration rate through cover soils and hydraulic conductivity.

Direct flux emissions from landfill surface

1. CH₄ and CO₂ fluxes were measured at the Robinson Deep and Marie Louise landfill sites using flux chambers placed across the landfill surface.
2. CH₄ and CO₂ emissions were generally low at both landfill sites, however emissions were higher at Marie Louise than Robinson Deep. CH₄ emissions at Robinson Deep peak at 0.1762 g/m²/day whilst the highest CH₄ emission measured at Marie Louise was 0.4414 g/m²/day. CO₂ emissions were generally higher than CH₄ emissions at both sites. This was also found by Fourie and Morris (2004) who measured CH₄ and CO₂ fluxes at four landfills in South Africa. The high CO₂ emissions relative to CH₄ emissions can be due to CH₄ oxidation in cover soils, however this was not determined in this study. Previous studies have shown that oxidation of CH₄ at some landfill sites can be between 30 and 100%.
3. The mean CH₄ flux of 0.0107 g/m²/day was measured from chambers placed at a closed section of Robinson Deep where a LFG extraction system was installed, reflective of the LFG extraction system aiding in decreasing emissions to the surface. Borjesson and Svensson (1997) also showed that LFG extraction systems decreased CH₄ emissions to the surface by as much

as 90%. Mean fluxes measured by Fourie and Morris (2004) at landfills in South Africa ranged between 12.7 g/m²/day and 56.5 g/m²/day (without LFG extraction).

4. Random sample CH₄ emission fluxes range between -0.0024 g/m²/day and 0.0338 g/m²/day, highlighting the spatial variability of CH₄ emissions across the landfill surface. Bogner *et al.*, (2011) shows CH₄ emissions can vary over 3-4 orders of magnitude during a single field campaign. Emissions can vary due to differences in waste amounts, age and composition in different parts of the landfill, hence emissions vary greatly over the surface over a few meters as described by Farquhar and Rovers (1973), Christensen and Kjeldsen (1996) and Morris (2001).
5. Flux emission rates measured during a passing of a cold front were higher than emissions measured under normal weather conditions. Maximum flux was measured at 0.1762 g/m²/day under normal weather conditions and 0.2579 g/m²/day during a cold front. This is consistent with previous studies. Cziepel *et al.*, (2003) show that a barometric pressure decrease of 10 hPa causes a tripling of the CH₄ emissions from a landfill. This is due to an increase in the CH₄ pressure gradient between the landfill and the atmosphere.
6. Fluxes are low when compared to CALMIM. The model shows high seasonal variability in surface emissions with oxidation and without where there are higher emissions in winter and lower emissions during the summer

with oxidation. Asadi *et al.*, (2013) showed similar results from measured surface emission fluxes at landfills in Australia. Emissions are comparable to modelled emissions associated with daily cover soils but are much lower than emissions with intermediate and final cover soils.

7. The low CH₄ and CO₂ surface fluxes could be attributed to the slow degradation of waste at the surface of the active cell. The waste degrades aerobically and reaches methanogenesis at a later stage or once another layer of waste is disposed above. High oxidation rates of CH₄ in cover soils to CO₂ also result in lower emissions to the atmosphere. Methanogenesis is reached deep within the landfill site as shown by the LFG extraction rates. A study measuring CH₄ and CO₂ fluxes at a municipal landfill site in Argentina found no CH₄ emissions and CO₂ emissions between 5 g/m²/day to 214 g/m²/day (in 50 sample points) and 31 g/m²/day to 331 g/m²/day (at 107 sampling points). The study found that the shallow waste placement and insufficient compaction of the waste resulted in aerobic waste decomposition and that waste degradation underwent anaerobic oxidation and never reached methanogenesis (Sanci and Panarello, 2012).

Polluting potential of landfill sites

The polluting potential of the landfill was assessed using the air dispersion model ADMS. Modelled concentrations are compared to the ambient air quality monitoring that was conducted at the site from May – August 2010.

1. H₂S, BTEX and PM modelled concentrations as well as ambient concentrations are within air quality guidelines and standards. However, the modelled dispersion shows the odour impact from the landfill to be significant.
2. There is an additional source from the north-west contributing to odours at the site as shown by the Conditional Bivariate Probability Function (CBPF) for H₂S which shows the highest probabilities of high concentrations (22 – 100 ppb) originate from the north-west.
3. BTEX and CH₄ CBPF shows the highest concentrations occur under very low wind speed conditions with a slight westerly origination indicative of emissions directly from the landfill site. There is also a contribution of a major source from the north-west.
4. Results show the landfill is a contributor to the odour issues in the area but that there are also other sources north-west of the landfill. The odour complaints could also be associated with freshly disposed waste that was not covered at the site. Hence, the odour pollution from the landfill is largely dependent on the manner in which the facility is managed.
5. To reduce odour nuisances, daily covering of waste with adequate cover material is necessary. Odours from stored leachate can be minimised by applying odour controls to the leachate pond. In the long term, the gas extraction and flaring would be effective in controlling gas emissions and odours from the landfill site.

Landfill gas extraction and application as Clean Development Mechanism projects in South Africa

Landfill gas (LFG) extraction and utilisation projects implemented in South Africa are assessed. Six projects in South Africa i.e. Ekurhuleni Metro Municipality, City of Johannesburg, two within eThekweni Metro Municipality, Chloorkop and Alton landfill sites are evaluated to gauge the applicability and success of each project under the Clean Development Mechanism (CDM) programme of the Kyoto Protocol.

1. The Ekurhuleni Metropolitan Municipality (EMM) established a LFG extraction and utilisation project at four of its landfill sites: - Weltevreden, Simmer and Jack, Rooikraal and Rietfontein. 650 973 tCO₂e has been reduced by the project since the start of operation in August 2007 and June 2015.
2. eThekweni LFG was the first LFG extraction and utilisation project to be implemented in South Africa. The project was implemented at three landfill sites: - Bisasar Road, Mariannhill and La Mercy landfills. 186 996 tCO₂e were generated at Mariannhill from project registration to December 2013 and 7 766 tCO₂e generated at La Mercy up to June 2009 when the project was abandoned.
3. The Alton landfill site operated for just over two years and produced approximately 3 603 tCO₂e. The site was not adequately maintained and

operated and hence was not successful in producing sufficient LFG to maintain an extraction and utilisation system.

4. The Chloorkop landfill site is privately owned by EnviroServ and to date is the most successful project implemented in the country. A total of 857 309.5 tCO₂e has been reduced by the project between January 2008 and July 2014 averaging at 10 852 tCO₂e per month. It has also undergone a number of successful verifications and has sold most of the emission reductions generated.
5. The Joburg landfill gas to energy project was awarded to EnerG Systems in 2007 which sees the private entity place all the capital and funding to purchase the equipment for the entire gas extraction system as well as engines for future electricity generation, at no cost to the municipality. In total the project has generated 205 084 tCO₂e of both VERs and CERs and have sold the bulk of the emission reductions.
6. Of the ten landfill sites in South Africa that have implemented LFG extraction and utilisation projects only eight are still operational and two are successfully generating electricity. All registered CDM projects except EMM have managed to sell the majority of the emission reductions generated by the projects.
7. Administrative issues have also delayed projects ranging from delays encountered with the completion of environmental impact assessments; delays in project registration with the United Nations Framework

Convention on Climate Change (UNFCCC) executive board; delays in appointing Designated Operational Entities (DOE's), transactional advisors and selling of carbon credits due to the Municipal Financial Management Act (MFMA) and supply chain policies within municipal entities are but a few of the challenges projects have experienced.

8. Despite the challenges the projects have had many sustainable development benefits for South Africa including job creation and transfer of specialised skills as well as mitigating close to 3 million tCO₂e.

Conclusions and recommendations

In conclusion the study has shown that waste disposal has a significant contribution to CH₄ emissions in South Africa and the majority of these emissions are in urbanised Metro Municipalities of the country. In comparison LFG emission simulations for LandGEM and IPCC Waste Model are very similar and are shown to over predict LFG generation when compared to actual gas yields at individual landfill sites. GasSim simulates much lower emissions and falls within actual gas yields at some sites. It is evident that a more accurate model is required to estimate LFG emissions in South Africa to assist in calculating the contribution of LFG emissions to national greenhouse gas emissions as well as to effectively determine the viability of LFG extraction and utilisation projects in the country.

However, despite the model results CH₄ emission fluxes from the landfill surface are fairly low and CO₂ flux emissions are higher than CH₄ emission. Flux measurements taken at a closed section where an LFG extraction system is installed are reflective of the LFG extraction system aiding in decreasing emissions to the surface. This study shows that CH₄ and CO₂ emission rates to the surface during winter are minimal from operational landfill sites in South Africa which have mostly daily cover soils. A previous study by Fourie and Morris (2004) also measured low emission flux rates and concluded that waste in semi-arid South Africa degrades slowly thus slowing down gas production within the landfill. However, this study shows that despite the semi-arid environment, sufficient gas is produced for the implementation of LFG extraction and utilisation projects from data obtained from CDM projects within the country. This differential results

between surface emission fluxes and gas generation can be due to a number of factors. The low CH₄ and CO₂ surface fluxes could be attributed to the slow degradation of waste at the surface of the active cell. The waste degrades aerobically and reaches methanogenesis at a later stage or once another layer of waste is disposed above. High oxidation rates of CH₄ in cover soils to CO₂ can also result in lower emissions to the atmosphere. Methanogenesis is reached deeper within the landfill site as shown by the LFG extraction rates. It is recommended that further studies to measure surface CH₄ and CO₂ emissions be conducted at a number of landfill sites in South Africa as well as to determine site-specific CH₄ oxidation rates to accurately determine the contribution of landfills to national greenhouse gas emissions.

As majority of the waste is disposed at landfill sites it is a great opportunity to mitigate emissions through the implementation of LFG extraction and utilisation projects. This study has shown that despite the predictions of LFG estimation models, emissions are highly dependent on a number of factors; including proper daily operations (cover and compaction) at the site, leachate production and waste composition. As described by Tchnobanoglous et al. (1993) and Pitchel, (1995) compaction of waste on a daily basis aids in LFG formation especially with dry waste as it increases the moisture content and reduces the intrusion of air within the landfill. In addition, a daily soil cover can also increase LFG production if the soil used provides important buffer capacity to the landfill allowing for optimum pH values for CH₄ production. El Fadel et al., (1995) highlighted that leachate production can inhibit methanogenesis, especially where domestic waste is co-disposed with industrial wastes. The presence of inhibitors such as heavy metals

has toxic effects on methanogenic bacteria and inhibits the formation of LFG (Tchnobanoglous et al., 1993; Pitchel, 2005). Lombard and Associates conducted a feasibility study for the Department of Energy (DME) which focused on testing the feasibility of using landfill sites for LFG extraction and utilisation for electricity generation in South Africa. Of the 453 landfills used in the study only 57 accepted enough waste volumes to justify being considered as a potential LFG to electricity project (DME, 2004). The study has shown that some projects have failed as a result of unfavourable operations during the disposal of waste at the landfill site resulting in insufficient gas production, highlighting the fact that LFG production is very much affected by the engineering and daily operations at the sites. Inadequate compaction of waste and cover can allow air to ingress and turn the site aerobic as in the case of Alton landfill site. The disposal of sludge and other leachate producing waste can cause gas wells to become blocked and corrode decreasing flow rates to below functional levels as in the case of La Mercy and Rietfontein landfill sites.

Capital costs, daily operational and maintenance expenditure as well as costs associated with the registration of projects, verification and transactional advisors for the sale of emission reductions currently make CDM projects financially unviable. If the current price for carbon credits increases beyond just a few cents and combined with electricity generation at a rate higher than current prices, CDM projects could become financially stable or profitable. This is a high possibility when the carbon tax comes into effect in South Africa.

Despite this, projects have developed new skills and created jobs in many sectors. They have aided in reducing the ambient pollution of landfill sites to the environment by reducing odours and have also aided in better operations at sites including proper cover and better overall management as this is vital for the LFG extraction system to operate optimally. Thus, the implementation of such projects should be carefully considered. LFG estimation models should be used with caution and trials for LFG extraction should be conducted for longer periods than the industry standard of 6 months, if possible.

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