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Pereira, Gabriel Philip Alves

2024
University of the Witwatersrand, Johannesburg
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Pereira, Gabriel Philip Alves. (2024). Unlocking Sustainability: An Exploration of the Potential for Food Waste Conversion into Energy and Value-Added Products for Effective Landfill Diversion in South Africa [Masters dissertation, University of the Witwatersrand, Johannesburg]. WIReDSpace. https://hdl.handle.net/10539/45411

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Unlocking Sustainability: An Exploration of the Potential for Food Waste Conversion into Energy and Value-Added Products for Effective Landfill Diversion in South Africa.



Gabriel Pereira

359729

A research report submitted to the Faculty of Engineering and the Built Environment, University of the Witwatersrand, Johannesburg, in partial fulfilment of the requirements for the degree of Master of Science in Engineering.

Johannesburg, 2023

DECLARATION

I, Gabriel Pereira, declare that this Research Report is my own, unaided work. It is being submitted for the Degree of Master of Science in Engineering at the University of the Witwatersrand, Johannesburg. It has not been submitted before for any degree or examination at any other University.

Gabriel Pereira

(Signature of candidate)

28 day of June 2024 in Johannesburg

ABSTRACT

This research report examines the potential for transforming food waste into energy and value-added products as a sustainable strategy for diverting waste from landfills in South Africa. The study uses a social cost-benefit framework to evaluate various waste conversion technologies, including anaerobic digestion, aerobic composting, vermicomposting, and black soldier fly larvae, within South Africa's unique socio-economic and environmental landscape. The study combines quantitative waste composition analyses with technology evaluations, revealing significant landfilling costs and underscoring the economic and environmental urgency of strategic food waste diversion. Highlighting gate fees' critical role, the findings advocate adjusting these fees to incentivise sustainable waste management practices. Notably, Black Soldier Fly larvae and Vermicomposting technologies emerge as dual-benefit strategies for waste reduction and resource recovery. The report calls for policy and infrastructure advancements to enable the broad adoption of these technologies, emphasising their importance in achieving sustainability and economic objectives.

ACKNOWLEDGEMENTS

Thank you to Professor Anne Fitchett for her guidance, patience, and constant motivation. Thank you to my friends, family, and Bryan for your motivation and support.

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LIST OF VARIABLES AND ABBREVIATIONS

AD - anaerobic digestion

AP - Acidification potential

AWT - Advanced waste treatment

BSF - Black soldier fly

DEA - Department of Environmental Affairs

DFFE - Department of Forestry, Fisheries and the Environment

DOC - Decay constant

ELCC - Environmental life-cycle costing

ECE - Environmental cost-effectiveness

EP - Eutrophication potential

EU - Energy use

GHG - Greenhouse gas

GWP - Global warming potential

INV - In-vessel composting

ISWM - Integrated solid waste management

LAP - Local air pollutants

LCA - Life cycle assessment

LCC - Life cycle costing

MSW - municipal solid waste

ODP - Ozone depletion potential

VC - Vermicomposting or vermiculture

WC - Windrow composting

CHAPTER 1: INTRODUCTION

This report critically examines the systemic inefficiencies in handling food waste in South Africa, a challenge that exacerbates the burden on already overloaded landfills, squanders the potential for sustainable energy creation, and neglects opportunities for producing value-added products. In the face of mounting environmental, social, and economic pressures, this study seeks to uncover viable pathways for transforming food waste into resources that contribute positively to society and the environment.

1.1 Background and Context

Globally, waste management presents a formidable challenge, characterised by the variable nature of waste generation and the complex systems of waste production and consumption. South Africa stands at a particularly critical juncture, with its major cities grappling with rapidly diminishing landfill capacity, further strained by inadequate planning and resources at various levels of governance (DEA, 2018; GreenCape, 2022b). Despite governmental aspirations towards a zero-waste landfill future, the prevailing lack of alternative waste disposal methods propels the continuous traditional discarding of waste.

The ethical dimensions of food waste are profound, touching on issues of equity, environmental stewardship, and economic justice. While vast amounts of food are wasted, hunger and food insecurity afflict millions globally, spotlighting the moral imperative to mitigate food waste (Gustavsson et al., 2011; FAO, 2013; Stenmarck et al., 2016; Schanes et al., 2018). In South Africa, the Department of Environmental Affairs (DEA) underscores the considerable economic loss tied to the disposal of recoverable resources, valued at R825 billion annually, a clear indication of the substantial economic and resource recovery opportunities being overlooked (DEA, 2016; Nahman et al., 2012; Oelofse & Nahman, 2013).

Reinforcing this perspective, the National Waste Management Strategy (NWMS) 2020 sets ambitious national targets for the diversion of organic waste from landfills, aiming to reduce organic and food waste to landfill by 40% and 30% respectively,

within five years in a bid to alleviate the environmental and social impacts associated with landfilling and incineration (DEFF, 2020). The urgency of addressing these issues is further magnified by the health risks associated with proximity to waste sites, as evidenced by recent studies (Mataloni et al., 2016; Dumcke et al., 2020; Tomita et al., 2020; Moolla et al., 2013).

The lack of compliance with municipal landfill regulations, the absence of data on the informal waste sector, and the dire realities faced by communities reliant on landfills for food and resources paint a bleak picture of the current state of waste management in South Africa (Klerk, 2023; Mepaiyeda, 2020). The alarming cost of edible food waste, amounting to R75 billion annually or 2.2% of the nation's GDP in 2013, underscores the critical need for comprehensive solutions that address waste management inefficiencies, optimise food waste utilisation, and ensure public health and compliance at landfill sites (De Lange & Nahman, 2015).

1.2 Statement of Problem

The current state of food waste management in South Africa presents significant challenges due to systemic inefficiencies that exacerbate the burden on already overloaded landfills, squander the potential for sustainable energy creation, and neglect opportunities for producing value-added products. Despite governmental aspirations towards a zero-waste landfill future, the lack of alternative waste disposal methods continues to propel the traditional discarding of waste. This situation is further complicated by the ethical, environmental, and economic dimensions of food waste, where substantial amounts of food are wasted while millions suffer from food insecurity.

1.3 Research Question

How can the conversion of food waste into energy and value-added products be optimised to reduce landfill usage and contribute to sustainable development in South Africa?

1.4 Aims

This research endeavours to critically assess the capacity of specific organic waste conversion technologies—namely anaerobic digestion, composting, vermicomposting, and Black Soldier Fly processing—in mitigating food waste challenges in South Africa. It aims to scrutinise the efficacy of these technologies in facilitating landfill diversion, enhancing energy production, and generating value-added products. By doing so, this study aspires to contribute to realising Sustainable Development Goal 12.3, aiming for a substantial reduction in food waste and loss. The research seeks to bolster food waste management practices through strategic recommendations, thereby diminishing food loss and waste within the South African context and offering insights into scalable solutions for broader environmental and economic sustainability.

1.5 Objectives

This research seeks to address the problem via the following objectives:

1. Assess the current state of food waste management in South Africa, including the primary sources, composition, and costs associated with landfilling.
2. Evaluate the potential of various organic waste conversion technologies for landfill diversion, energy production, and value-added product creation within South Africa's context.
3. Formulate strategic recommendations for policy and regulatory measures to enhance the adoption of sustainable food waste management technologies in South Africa.

1.6 Structure of report

The report is structured to systematically explore the potential of food waste conversion into energy and value-added products in South Africa. It begins with **Chapter 1**, an introduction that sets the stage for the investigation, highlighting the problem of food waste management. **Chapter 2**, the literature review, provides a comprehensive overview of food waste generation, the cost of landfilling, and various diversion strategies. **Chapter 3** discusses the challenges of landfill compliance and

the state of policy in waste management governance. Methodological details are outlined in **Chapter 4**, focusing on evaluating waste conversion technologies. The subsequent chapters delve into the cost of landfilling, offering a nuanced analysis through different models and methods. **Chapter 5** determines the cost of landfilling in South Africa using the benefit transfer method and reviewing local literature. A comparative cost-benefit analysis of several waste management technologies follows in **Chapter 6**, assessing their viability. The results in **Chapter 7** present critical findings on landfilling practices, technology comparison, and a sensitivity analysis. The discussion synthesises these findings in **Chapter 8**, examining the status quo, policy implications, and the benefits of alternative technologies. The report concludes with **Chapter 9**, a summary of insights and recommendations for future waste management strategies in South Africa.

CHAPTER 2: LITERATURE REVIEW

2.1 Overview of Food Waste Generation in South Africa

It has been reported that approximately one-third of all food generated worldwide for human consumption gets lost or wasted (Gustavsson et al., 2011). This wastage is largely a product of inefficiencies in the food value chain and results in considerable economic, social, and environmental impacts, cumulatively amounting to a cost of US$940 billion (Hanson & Mitchell, 2017). In the context of South Africa, the total food loss and waste were estimated to be about 9.04 million tonnes per year on average (for the period between 2007 and 2009), excluding the consideration of food imports and exports. This value represents 31.4% of the country's average yearly agricultural output, around 28.79 million tonnes (Oelofse & Nahman, 2013). Upon including the data on imports and exports, the overall estimate of food waste increases to 10.2 million tonnes annually, inflicting a societal cost of R61.5 billion per annum, equivalent to 2.1% of South Africa's GDP (Nahman & de Lange, 2013; Oelofse et al., 2021).

Food waste in the below data by Oelofse et al. (2021) refers to the discarding or loss of food that is still suitable for human consumption and does not include the inedible portion (peelings, bones, etc.). It can be found at various stages of the food supply chain, from production and processing to distribution, retail, and consumption. The below Figure 1 describes the distribution at different stages of the value chain.

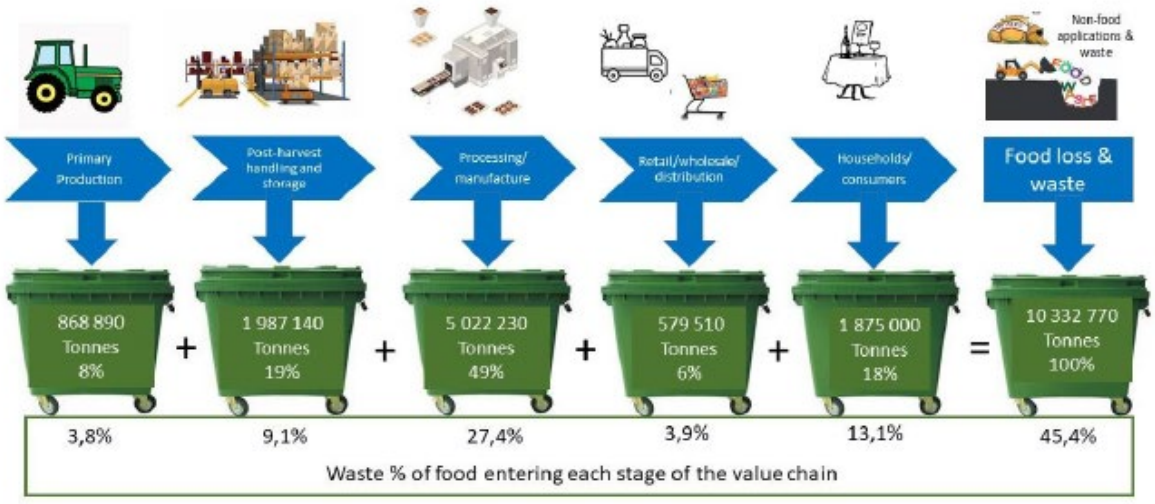

Figure 1: Average annual food losses and waste along the food value chain in South Africa for the period 2014-2018 (Oelofse et al., 2021)

2.1.1 Data on Edible Fraction of Food Waste

The Waste RDI Roadmap's 2021 report, curated by Oelofse et al., significantly advanced the understanding of food loss and waste in South Africa by providing reliable, science-backed data. While the report's findings support the general methodology previously used for estimating food loss and waste, they highlight a significant oversight: the non-food by-products derived from food losses and waste.

The report applied updated assumptions to calculate food loss and waste percentages at each stage of the food value chain. Based on this refreshed approach, pre-COVID-19 food losses and waste in South Africa were estimated at 10.3 million tonnes per annum, using the five-year average food supply data (2014-2018). This estimation aligns closely with the 2013 figures, although the allocation of losses and waste across the value chain has evolved.

Nearly half (49%) of the losses and waste occur at the processing and packaging stage. In contrast, food waste at the consumption stage, representing 18% of the total, is over three times higher than prior South African estimates (Oelofse et al., 2021).

Table 1: Extracted directly from the report Oelofse et al. (2021), outlines outlining the annual averages of food waste across South Africa's value chain, based on the food supply average over the 2014-2018 period (in thousands of tonnes).

Commodity Group	Average annual food production	Post harvest handling and storage	Process and packaging	Distribution	Consumption	Total
Cereals	102.66	589.45	3494.33	121.58	863.85	5171.87
Roots and Tubers	181.8	160.35	132.83	30.89	19.68	525.55
Oilseed and Pulses	3.04	115.64	111.31	1.48	12.36	243.83
Fruit and Vegetables	335.74	621.22	381.24	131.57	463.43	1933.2
Meat	215.68	175.45	0	223	296.27	910.4
Fish and seafood	2.75	1.6	119.85	16.96	4.51	145.67
Milk	27.22	323.42	782.68	54.03	214.91	1402.26
Total	**868.89**	**1987.13**	**5022.24**	**579.51**	**1875.01**	**10,332.78**
Waste % of food entering each stage of value chain	3.81%	9.07%	27.35%	3.89%	13.09%	

This updated estimation of 10.3 million tonnes of annual food losses and waste corresponds to 34.3% of local production or 45.4% of the available food supply entering the country's food value chain. While the overall magnitude of the estimate aligns with previous studies by Nahman & de Lange (2013) and Oelofse & Nahman (2013), it differs in terms of the losses and waste distribution across the value chain and among commodity groups.

Table 2: Percentage of food losses at each stage of the value chain between studies (Oelofse et al., 2021)

Stage in the value chain	Oelofse and Nahman (2013)	Nahman & De Lange (2013)	Oelofse et al. (2021)
Production	26	26	8
Post-harvesting handling and storage	26	24	19
Processing and packaging	27	25	49
Distribution	17	20	6
Consumption	4	5	18

As seen in Table 2 above, a considerable portion of the losses (82%) occur pre-consumer. However, there is a substantial increase in waste generated during consumption, rising from 5% (Oelofse & Nahman, 2013) to 18%. This surge is concerning as consumption stage waste is typically high-value and challenging to repurpose, despite often being preventable. Mitigating consumption stage waste is a complex task as it necessitates alterations in human perceptions and behaviours.

2.1.2 Estimates of the Inedible Fraction of Food Waste

Using data from the UK as cited by Nahman et al. (2012), it is observed that 19% of consumer-discarded food is inedible, leaving 81% still consumable. Lacking specific data for South Africa, this study used a similar 81% edible food waste ratio for the analysis, considering 81% of the total household food waste generated in the country. This assumption was consistent in Nahman and De Lange's 2013 analysis, which explored food losses across the value chain, though their data specifically centred on edible food waste. Oelofse et al. 2021 also adopted this 80:20 ratio, and it will be utilised to determine the inedible food waste segment, following the approach outlined in De Lange & Nahman, 2015.

An estimated 10.3 million tonnes of edible food waste is generated across the value chain annually (see distribution in Table 1). Given that this represents 81% of the total food waste, the entire food waste volume in South Africa is approximately 12.76 million tonnes per year. This volume breaks down to 10.3 million tonnes being edible and 2.4 million tonnes inedible. This paper will analyse the full scope of food waste, both consumable and non-consumable, using the term 'food waste' to denote the collective volume.

2.1.3 Estimate of Food Waste Fraction Landfilled

This report will focus on food waste disposed of in general waste landfills. General waste includes waste generated from households, offices, schools, restaurants, and other sources that are not hazardous. It is defined by the Waste Amendment Act of 2014 as:

> “waste that does not pose an immediate hazard or threat to health or to the environment…” (NEMA, 2014)

The below Table 3 defines the general waste categories in more detail:

Table 3: General waste by category adapted from the 2018 SoWR

Waste Type	Total waste managed (tonnes)	Percent disposed (%)	Total volume disposed (tonnes)	Percentage of disposed waste (%)
Municipal Waste	4,821,430	100	4,821,430	13.33
Commercial Waste	360,884	100	360,884	1.00
Fly ash	4,346,080	96.9	4,211,352	11.65
Bottom dust	6,489,080	96.9	6,287,919	17.39
Slag	4,859,025	100	4,859,025	13.44
Organic Waste	19,251,600	50.8	9,779,813	27.04
Construction waste	4,482,992	48	2,151,836	5.95
Paper	2,139,706	42	898,677	2.48
Plastic	1,099,254	56.3	618,880	1.71
Glass	2,791,003	28.8	803,809	2.22
Metals	3,533,059	20	706,612	1.95
Tyres	162,167	0	-	0.00
Other	729,615	91	663,950	1.84
Total	**55,065,895**		**36,164,185**	**100%**

The below, Figure 2 adapted from the 2018 SOW report, report has highlighted organic waste as the bulk of general waste going to landfill. This waste stream has been flagged as offering significant opportunities for reducing the volume of waste landfilled. Greater emphasis needs to be placed on this waste stream as a significant amount of waste can be diverted from landfill. The 2018 SoWR data, summarised in Table 3 above, suggests that on average 27% of the total volume of waste landfilled is organic waste.

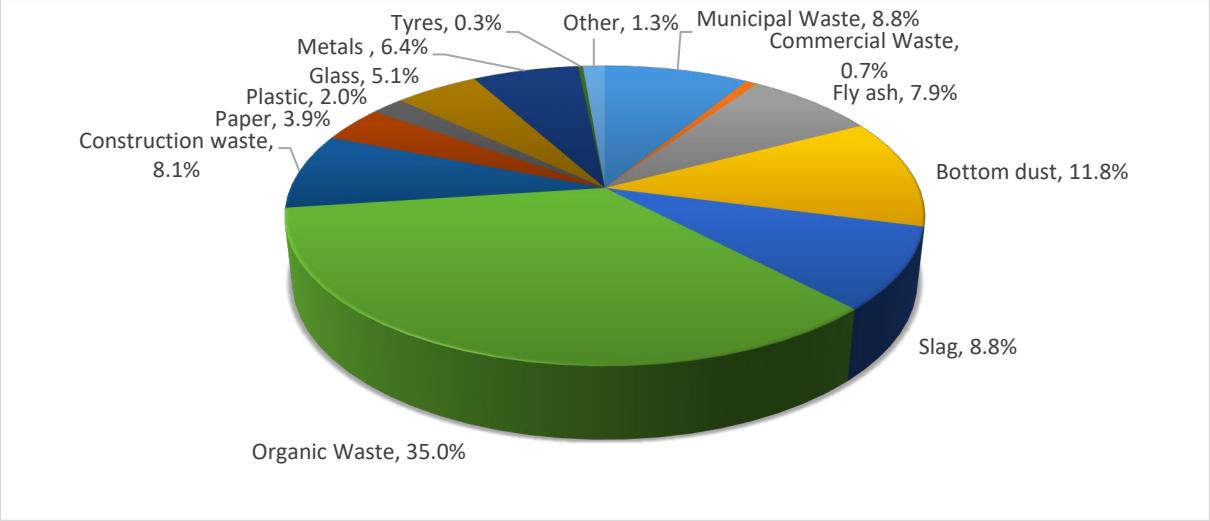

Figure 2: Breakdown of general waste generated – adapted from 2018 SoWR

From Table 3 that the total organic waste fraction is broken down as follows:

Total organic waste generated locally	= 19,247,851tonnes
Total organic waste managed	= 19,251,600 tonnes
Total disposed (50.8%)	= 9,779,812.8 tonnes

Total organic waste disposed of is approximately 9.8 million tonnes or 27% of the total volume of waste disposed. It is assumed that this is sent to landfill as it has not been recycled or diverted which has been accounted for. The 2018 SoWR notes the difficulty in assessing the constituents of this organic fraction accurately, but it is made up of the following categories:

- Garden Waste in the municipal solid waste stream.
- Food waste in the municipal solid waste stream.
- Wood waste from sugar mills, wood sawmills and paper and pulp mills.

Waste from the harvesting of food crops, sugar, and logging is typically left in the field for soil enhancement or as animal feed and is thus not included in this analysis (Oelofse et al., 2021; DEA, 2018). The State of Waste Report (SoWR) defines food waste as stemming from kitchens, restaurants, and the processes of food preparation, distribution, and retail. Agricultural harvest waste is excluded, given its common use for soil improvement or as animal feed. Leveraging waste characterisation studies, the proportion of food waste in the municipal waste stream

can be discerned. In 2017, food waste comprised around 10.8% (2.5 million tonnes) of the country's municipal waste.

Recent 2021 research by Oelofse et al. estimates edible food waste at 7.5 million tonnes during the distribution and consumption stages. This data highlights that recorded landfill volumes don't fully represent the diversion of food losses to non-food derivatives. Field data suggests farm waste is often repurposed as animal feed, reintroduced as soil nourishment, or used for compost. Excess food is usually donated, and food losses during processing get redirected to non-food uses, such as animal feed or oil extraction, enzymes or other components (Oelofse et al., 2021).

The landfill food waste primarily originates from distribution and consumption, totalling approximately 2,454,520 million tonnes (Oelofse et al., 2021). This aligns with data from IWMPs and waste characteristic studies referenced in the SoWR. About 25% of the discarded organic waste (9,779,812.8 tonnes) is believed to be food waste from these supply chain stages (Oelofse et al., 2021). This 25% represents only the edible fraction; an additional 6% accounts for the inedible fraction, as detailed in the previous section on inedible food waste estimates. A comprehensive data overview can be found in Table 4.

Table 4: Summarizing the proportion of total food waste landfilled.

Total tonnage disposed to landfill (tonnes per annum)	**36,284,479.84**
Edible food waste fraction (tonnes per annum)	2,454,520.00
Inedible food waste fraction (tonnes per annum)	575,751.60
Total food waste (tonnes per annum)	3,030,271.60
Organic fraction landfilled (tonnes per annum)	9,779,812.80
Total food waste as % of organic waste fraction	31%
Total food waste as % of total waste	8.35%

The 2020 Greencape Market Intelligence report on Waste in the Western Cape suggests that 14.5% of waste was from food waste in mixed forms, with residual organic at 5.9% and garden waste at 7.4%. The total organic fraction is 28.19%. This reinforces the above assumption that 27% of waste landfilled is organic waste.

2.2 Determining the cost of landfilling food waste in South Africa

This report aims to determine the current cost of food waste management to compare with viable alternatives. This is not just the financial costs, capital and operation expenditure but also includes the cost to the environment as well as surrounding communities (Nahman, 2011). The full cost will be determined in detail in Chapter 5 of this report, but a baseline of the status quo needs to be determined from the literature. This section intends to review the available literature on external landfilling costs in South Africa and will require a determination of the current landfill practices, levels of compliance, and value derived from food waste to get a baseline assessment.

2.2.1 Methods to determine the cost of managing food waste

In evaluating waste processing technologies, researchers utilise a blend of methods to appraise their economic and social costs, thereby guiding the advancement of these technologies. Since the 1970s, both qualitative and quantitative methods have been instrumental in decision-making concerning waste disposal alternatives (Song et al., 2020). Analytic hierarchy process (AHP) and multicriteria decision model (MCDM) represent qualitative methodologies that assess various influencing factors of these technologies, ranking them against other waste management options based on expert evaluation and historical data. These models consider a spectrum of criteria, including cost, demand, waste levels, and environmental impact, with the selection and integration of these criteria being pivotal to the process (Medina-Mijangos & Seguí-Amórtegui, 2020a).

On the quantitative side, methodologies such as life cycle assessment (LCA), social cost–benefit analysis (SCBA), and optimisation models are employed. These approaches aim to monetise abstract social and environmental costs using metrics like shadow price, willingness to pay (WTP), or carbon prices, thereby providing a holistic economic valuation of waste management technologies and guiding investment decisions (Song et al., 2020).

LCA, in particular, has been a vital tool for assessing the environmental impacts of waste treatment processes, analysing the inputs, outputs, and potential

environmental impacts throughout a product's life cycle. Friedrich & Trois (2013a) utilised LCA to compare greenhouse gas emissions and pollution from a variety of waste management options and traditional landfilling methods in South Africa. SCBA, along with sustainable return on investment (SROI), evaluates both environmental and social effects, pricing impacts uniformly to assist decision-makers in understanding the net social effects of waste management. SCBA encompasses investment and operational costs, as well as social and environmental impacts, while SROI focuses on sustainability metrics like net present value (NPV) and payback period, incorporating risk analysis for a more comprehensive assessment (Song et al., 2020).

To effectively assess various waste management technologies within the South African context, this study employs a methodology developed by Medina-Mijangos et al. (2020) for the technical-economic analysis of municipal solid waste systems. This approach, rooted in social cost-benefit analysis, aims to reduce uncertainty and provides a robust tool for evaluating the viability of waste management infrastructure projects, taking into account economic, social, and environmental dimensions. It's adaptable to various municipal solid waste (MSW) management infrastructures and waste streams and follows a seven-step process encompassing objective definition, stakeholder identification, financial analysis, and sensitivity analysis (Medina-Mijangos et al., 2020b).

2.2.2 Social cost -benefit analysis

Social Cost-Benefit Analysis (SCBA) translates the environmental consequences of Municipal Solid Waste (MSW) management into monetary values, integrating these with direct costs to determine the comprehensive societal expense, also known as life-cycle cost-benefit analysis (Yang et al., 2022). This process involves assigning monetary values to the effects of pollutants on health, the environment, and quality of life. Specifically, the analysis calculates the external costs of environmental impacts by multiplying the emission quantities by their respective monetary values (Song et al., 2020). It also considers the external costs associated with land use and the negative impacts of waste treatment facilities (Nahman, 2011). While the methodology for quantifying air emission impacts is relatively established, methods

for assessing water emissions' financial impacts require further development. The research in this area primarily originates from Europe and the USA, suggesting a need for localised monetary valuations to reflect different economic contexts. Yang et al. (2022) emphasises that while air emissions have been the primary focus of external cost assessments in MSW management, other factors like water and soil pollution, resource use, and social effects are less frequently included but are equally important for a comprehensive evaluation.

Life Cycle Assessment (LCA) is commonly utilised to quantify environmental benefits, but its integration into Social Cost-Benefit Analysis (SCBA) involves distinct steps across methodologies. SCBA employs LCA to quantify pollutant emissions, which are then translated into externality costs and folded into economic assessments.

2.2.3 Current food waste management practices in South Africa

Landfilling remains the primary waste disposal method in South Africa, responsible for approximately 80% of the country's waste management as of 2017 (DEA, 2018). Out of over 800 landfill sites sprawled across the nation, a concerning 80% fall short of required regulations, thus undermining South Africa's waste management strategies (DEA, 2012). This report looks at the portion of food waste being sent to general waste landfills and how that fraction is currently being treated.

The implications of landfilling extend beyond mere waste disposal; they pose significant threats to both environmental sustainability and public health. Contamination of air and water, soil degradation, and the release of potent greenhouse gases like methane from decomposing food waste underscore the environmental impact of landfills (Eshet et al., 2006). Furthermore, unchecked landfills risk contaminating water sources, potentially leading to the spread of diseases (Nyika et al., 2021). Though financially cost-effective in comparison to other waste management strategies, such as recycling or waste-to-energy, the continuation of landfilling by many municipalities does not take the full cost into account, including costs incurred to the environment and community. The potential sustainability issues raised by this method are hard to ignore, given the reduced

space for landfills and the mounting recognition of their environmental and health consequences (DEA, 2018).

Disturbingly, of the waste produced, a staggering 65%—equivalent to roughly 23 million tonnes—could be recycled or utilised for energy. However, only about 34% of the waste is currently being recycled or used for energy (Nahman, 2011). This underutilisation not only represents a significant loss of reclaimable resources but also puts immense financial, socio-economic, and environmental burdens on the nation. With urban areas running out of land, the urgency for reformed waste management practices grows.

For those living or working near landfill sites, the risks are palpable. Such sites, often noisy, malodorous, and aesthetically displeasing, pose tangible threats. Beyond the risks posed by waste collection vehicles, air pollution from spontaneous combustion and fires, water and soil contamination, and health hazards such as respiratory diseases, cancers, and birth defects are all real concerns. Alarmingly, a 2023 audit revealed that a mere 17.5% of the 161 assessed landfill sites complied with 80% or more of the set standards, highlighting severe lapses in waste management at a national scale (De Klerk, 2023).

The financial and societal costs of landfilling cannot be overstated. While the direct financial expenses encompass the costs of site establishment, operation, and eventual closure, the external or indirect costs affect the broader community. These externalities, or societal side effects not borne by those financially invested in the landfill, permeate various facets of society and even future generations. Notably, landfills generate landfill gas (LFG) and leachate from decomposing wastes. These emissions, whether to air, soil, or water, harm human and ecological health alike. Additionally, the nuisances associated with landfills—noise, odour, litter, pests, dust, and increased traffic—degrade the quality of life for those in the vicinity (Nahman, 2011).

While there are other external costs, such as the opportunity costs tied to the land occupied by the landfill, emissions and nuisance remain the predominant concerns when evaluating the adverse external impacts of landfilling (BDA Group, 2009).

Table 5: Externalities associated with landfill disposal (Nahman, 2011)

Category	Example of externalities	Source	Major Impacts
Emissions	Methane (CH_4) Carbon Dioxide Local air pollutants (e.g. sulphur dioxide, nitrogen oxides, particulates and volatile organic compounds) Pollution displacement (Positive benefit) Emissions to soil and water	LFG LFG, transport, fires LFG, transport, fires Energy recovery (LFG capture) Leachate	Global warming Global warming Human health, damage to buildings, crops, and the environment. Avoid global warming and human health. Human and environmental health
Disamenity	Odours, dust, wind-blown litter, vermin (rodents, flies, carrion birds), noise, traffic/congestion, visual intrusion, elevated fire risk.	Proximity to landfill site, transport, methane and LFG emissions, fires.	Nuisance/unpleasantness.

2.2.4 Compliance

In 2014, AfriForum initiated the #CleanSA campaign, aiming to improve waste management across South Africa by fostering collaboration between communities, the Department of Forestry, Fisheries and the Environment (DFFE), provincial departments, and local municipalities. The most crucial of these tiers from a waste management standpoint is the local municipalities, given their direct connection with communities (De Klerk, 2023).

Subsequently, AfriForum introduced its landfill site audit report. The objective of this report was to assess the compliance of both legal and illegal landfill sites across the 160 municipalities where AfriForum has branches. The audit scrutinised the alignment of these sites with minimum requirements and compared them to their waste management licenses. AfriForum stands out as the sole entity publishing such data about South African landfills and believes that many municipalities are not adhering to waste regulations due to a lack of accountability (De Klerk, 2023).

The 2023 Municipal Landfill Audit Report delved into waste management practices within select municipalities. It evaluated if they were upholding recommendations for best practices and meeting environmental, health, and safety standards. The compliance of each municipality was represented as a score out of 100 (De Klerk, 2023). For a more comprehensive assessment in 2023, AfriForum engaged with institutions like the CSIR and IWMSA, incorporating their insights to gather specific data, including estimations about the expected lifespan of certain landfill sites (De Klerk, 2023).

Table 6: Statistics pulled from the AfriForum 2023 Municipal Landfill Audit Report for the 2023 year only, per province. (De Klerk, 2023)

Province	Number of landfill sites audited by AfriForum	Number of landfill sites that complied with more than 80% of the minimum requirements	Number of landfill sites that did not comply with the minimum requirements	Number of landfill sites used in this study
	2023	2023	2023	2023
Gauteng	15	3	12	19
KwaZulu-Natal	10	1	9	10
Limpopo	14	2	12	14
Mpumalanga	22	1	21	20
Northern Cape	17	0	17	17
Northwest	23	5	18	21
Eastern Cape	8	2	6	8
Freestate	27	0	27	26
Western Cape	25	14	11	23
National total	**161**	**28**	**133**	**158**

Table 6 offers an overview of the data from 2023. Out of the 161 audited landfill sites in 2023, just 28 (or 17.5%) met or exceeded 80% of the basic criteria for such sites. Consequently, 133 municipal landfill sites (or 82.5%) failed to adhere to these standards. This data underscores significant gaps in the waste management systems and the accountability of personnel throughout the nation (De Klerk, 2023).

Figure 3 gives a provincial and national overview of landfill compliance. Most provinces have shown a steady decline in compliance since 2014, with only the Western Cape and Gauteng maintaining a relatively high compliance level. Nationally, the compliance score for 2023 is 17.5% compliance (De Klerk, 2023).

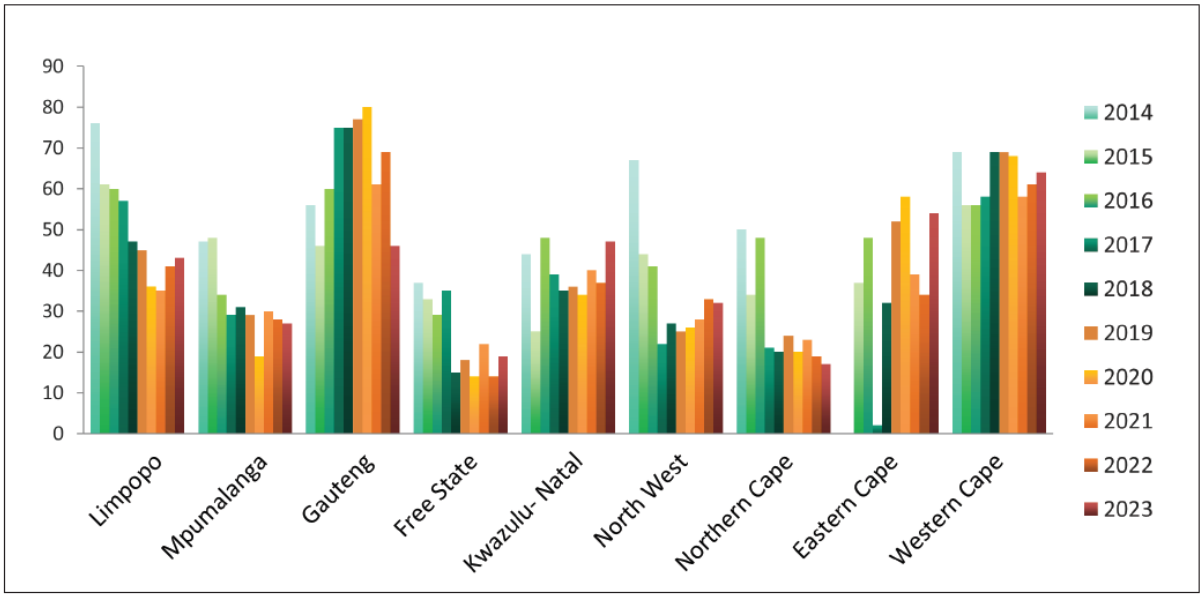

Figure 3: 2023 Municipal Landfill Audit Report, indicating the average audit scores for each province from 2014 to 2023 (De Klerk, 2023).

2.2.5 Waste-to-Energy

Waste-to-Energy (WtE) is a burgeoning waste management strategy that facilitates the transformation of waste into energy, primarily as electricity or heat. Its potential for reducing landfill waste volumes and fostering renewable energy generation has initiated several projects, such as landfill gas-to-electricity, biomass-to-energy, and refuse-derived fuel (RDF) projects in South Africa (DEA, 2018). However, South Africa's WtE sector, though promising, is still in the early stages, impeded by considerable challenges like high initial costs, technical difficulties, and the lack of a supporting policy and regulatory structure (DEA, 2018).

In a nation where over 60% of municipal waste is collected, South Africa is witnessing several burgeoning WtE projects (Coelho et al., 2019). Notably, Cape Town city utilises the organic fraction of municipal solid waste (MSW) for biogas production, supplying approximately 600 GJ/year of compressed biomethane, a viable substitute for industrial natural gas (Coelho et al., 2019). Concurrently, Johannesburg employs sewage for biogas production, facilitating combined heat and power (CHP) for onsite wastewater treatment at municipal wastewater treatment plants. Furthermore, landfill gas (LFG) is captured for electricity generation, ranging between 1 and 6 MW at multiple landfill sites in Durban and Johannesburg (Mbazima et al., 2022). A noteworthy projection is that the organic waste from the six largest

city municipalities in South Africa could potentially generate 207MW of electricity (SACN, 2014).

The organic fraction of municipal solid waste (OFMSW), which comprises approximately 15%–35% of the total waste, can be utilised for energy production. Annually, it is estimated that around 244,896–571,424 tonnes of OFMSW are available, and with an energy value of 9MJ/kg, an estimated 2204 TJ per annum is theoretically achievable. However, it is anticipated that only around 35%–40% of this will likely be available for anaerobic digestion to yield methane, resulting in an estimated 841 TJ per annum (Mbazima et al., 2022).

Promising landfill gas-to-electricity (LFG-E) projects are also thriving in other South African cities. For example, Durban hosts two successful landfill gas-to-electricity projects at Marianhill (1MW) and Bissasar Rd. (6.5MW). Moreover, there's potential for energy generation from sewage, as seen in the upgrades to municipal wastewater treatment works in Johannesburg, which capture energy from sewage while assisting with wastewater treatment (Mbazima et al., 2022).

Noteworthy is the biogas WtE plant in Athlone Industria, Cape Town, commissioned by New Horizons Energy in 2017. This plant receives around 643 tonnes of waste per day, with recycling and organic waste extraction achieving an 88% diversion rate from landfill. It generates approximately 670m^3 per day of biomethane, equivalent to a 5MW biogas-to-electricity production annually (Mbazima et al., 2022).

While these initiatives are promising, they underscore the necessity of increased investment, supportive policies, and technological advancements to fully exploit the potential of WtE in South Africa. With the right commitment and effort, the benefits of WtE can go beyond energy production, reducing landfill usage, mitigating greenhouse gas emissions, and supporting a circular economy (Coelho, 2019; SACN, 2014).

Table 7 indicates provinces with known WtE projects. There may be additional private sector projects not taken into account, but based on the below, 19.5MW is

currently the planned capacity nationally. A total potential capacity of 60.2MW is forecast based on current known municipal projects.

Table 7: Known Waste to Energy (WtE) projects in South Africa. (Mbazima et al., 2022)

Province	Site Name	WtE Technology	Planned Capacity (MW)	Landfill Status
Kwa-Zulu Natal	Marian Hill	LFG-to-electricity	1MW	Closed
Kwa-Zulu Natal	Bissasar Rd.	LFG-to-electricity	6MW	Closed
Kwa-Zulu Natal	Mariannhill	LFG-to-electricity	1MW	Closed
Western Cape	New Horizons	Anaerobic Digestion	5MW	NA
Western Cape	Coastal Park	LFG collection only	4MW (potential)	Operational
Western Cape	Bellville South	LFG collection only	4MW (potential)	Closed
Western Cape	Vissershok	LFG collection only	2-7MW (potential)	Operational
Western Cape	GLM	Incineration	5MW (potential)	Not operational
Western Cape	DLM	LFGE – EIA stage	10MW (potential)	Operational
Western Cape	SLM	LFGE – EIA stage	1.6MW (potential)	Operational
Gauteng	Simmer and Jack	LFG-to-electricity	1MW	Operational
Gauteng	Rietfontein	LFG collection only	Doubtful	Operational
Gauteng	Rooikraal	LFG collection only	1MW (planned)	Closed
Gauteng	Welteverden	LFG collection only	2MW (planned)	Operational
Gauteng	Robinson Deep	LFG-to-electricity	5.5MW	Operational
Gauteng	Ennerdale	LFG collection only	0.5MW (planned)	Operational
Gauteng	Goud koppies	LFG collection only	3.3MW (planned)	Operational
Gauteng	Linbro Park	LFG collection only	3.3MW (planned)	Closed
Gauteng	Marie Louise	LFG collection only	6MW (planned)	Operational

2.2.6 Benefit transfer method

Two pivotal studies that evaluated the cost of landfilling per tonne have been undertaken; one assessed the environmental and social costs (Nahman, 2011), and the other, the financial cost (De Wit & Nahman, 2009). These findings were later synthesised by Nahman et al. (2012) to estimate the total landfilling cost.

The disposal of food waste into landfills represents a lost resource, both as high-value edible food waste and lost opportunity cost. Further, it induces costs associated with the unnecessary disposal of organic waste and the associated costs of landfilling. These costs extend beyond finances and involve 'externalities,' or negative social and environmental impacts (Nahman, 2011). The decomposition of organic waste results in landfill gas (LFG) and leachate, both of which negatively impact human health, the global climate, and ecological health. Additional externalities include the consequences of transporting waste, such as air emissions,

accidents, congestion, and the 'nuisances' associated with living near landfill sites (noise, odour, litter, vermin, dust, etc.) (Eshet et al., 2006).

Nahman (2011) estimated the external cost per tonne of municipal solid waste entering Cape Town's landfill sites for LFG and leachate emissions, transport externalities, and disamenity (such as odours and pests). The external costs of landfill gas (LFG) and leachate emissions were determined using the benefits transfer method, which involves drawing on existing valuation estimates from other studies and applying them to the study site in question, making appropriate adjustments for differences between the original study site(s) and the new study site. Adjustments included for social biases, climate, and waste composition (De Lange & Nahman, 2015).

To evaluate the external costs of existing landfills across South Africa, the external costs for the following types of landfills are to be determined:

1. Urban and rural landfills without energy recovery (majority in South Africa).
2. Urban and rural landfills with LFG collection and flaring.
3. Urban and rural landfills with energy recovery.

The benefit transfer method is applied by translating existing studies to the study area of interest (Nahman, 2011). The full method is described in section 5.1 Benefit Transfer Method. Studies have looked at the costs of individual impacts of various pollutants, which can be applied to the estimated emissions of each landfill site.

There is a variety of methodologies that can be used to determine estimates of external costs of landfilling; these include but are not limited to market-based techniques (productivity method, human capital method, repair cost method), revealed preference techniques (travel cost method, hedonistic price method, proxy good), and stated preference method (contingent valuation, choice modelling) (BDA Group, 2009). Each method has different levels of data requirements, reliability and applicability and aims to estimate the damage costs associated with the health of environmental outcomes and determine the required compensation or willingness of surrounding communities to accept such damages (European Commission, 2000).

It is key, however, to identify potential impacts from actual impacts. While individual pollutants may be extremely harmful to human health or the surrounding environment, the proximity to people or potential discharge to the surrounding environment needs to be determined. The actual impact may be relatively small depending on the context; for example, air emissions in crowded urban areas will have higher impacts than in rural areas. It is also important to evaluate the literature being adapted using the benefit transfer method to ensure that costs are applicable to the site being studied (BDA Group, 2009).

Table 8: Summarising literature on landfill damage costs for different pollutants associated with landfilling.

	Country	External Cost (per tonne waste)	Assumed damage cost per tonne of waste (2002 ZAR)	Reference
GHG – modern	Europe (unclassified)	5€	155.14	(European Commission, 2000) – from a collective of studies in the EU and US
GHG – old		8€	253.87	
LAP – modern		0.1€	2.82	
LAP – old		0€	0	
Leachate – modern		0€	0	
Leachate – old		1.50€	6.2-12	
Disamenity – modern		10€	41.69	
Disamenity – new		10€	41.69	
Pollution displacement		-4€	-125	
GHG – Urban	Australia	11 AUS	155.14	(Zero Waste SA, 2006)
GHG – Rural		6 AUS	84.62	
LAP – Urban		0.4AUS	5.64	
LAP – Rural		0.3AUS	4.23	
Leachate		0.0004-0.002	0.04	
Disamenity – rural		0AUS	1.4-14	
Disamenity - urban		5AUS	2.8-28.05	
GHG – no collection	US	20AUS	282.07	(Miranda & Hale, 1997)
GHG – flaring		7AUS	98.73	
LAP – low		10AUS	141.04	
LAP – high		25AUS	352.59	
Leachate		0-2AUS	0-28.21	
GHG – low	UK	5AUS	70.52	(Davies & Doble, 2004)
GHG – high		17AUS	239.76	
Leachate - range		0-2AUS	28.21	
Disamenity	UK	7-9AUS	98.73-126.93	(DERFA, 2003)
Disamenity	Greece Spain	1.38 Eu/t-2.75 E/t 1.27e/t-2.54e/t	7.22-14.4 6.95-13.91	(Eunomia, 2002)
Disamenity	South Africa	R0.00-57.46	0-105.23	(Nahman, 2011)
Disamenity	South Africa	R5.83	11.86	(Du Preez & Lottering, 2009)

Table 8 Summarises some of the literature on landfill external or damage costs associated with pollutants or disamenities. The costs are adjusted for the year and currency to 2022 ZAR. However, further adjustment may be required for site-specific factors like socioeconomic factors, climate, and waste composition. Most studies listed appear in Europe and the UK, which have different climactic conditions from those spread across South Africa. Further, many studies have included data from other regions, like the US, and adjusted them accordingly.

The BDA report, as referenced by Nahman (2011), offers insights most relevant to South Africa's scenario, examining emission factors from landfilling across different climate categories akin to those in South Africa. This report's breadth, covering varied landfill management practices, locations, and climatic conditions, closely aligns with South African environments and population proximities. Additionally, it considers the substitution of coal-fired electricity with electricity generated from landfill gas, mirroring South Africa's energy context.

2.3 Food Waste Diversion Strategies – Best available technologies in developing countries

The landscape of food waste management is evolving, with a focus on innovative and sustainable technologies that cater to the unique challenges faced by developing countries. Among these, alternative waste treatments (AWT) are gaining prominence, leveraging biological processes, bio digestion, biomethane production, and insect protein conversion to mitigate environmental impacts. These AWTs stand out for their lower installation and operational costs, making them viable solutions for regions with limited resources and infrastructure (Coelho et al., 2019).

The pursuit of efficient waste management transcends mere financial considerations, encompassing a broader spectrum of economic and societal benefits. The 2015 Global Waste Management Outlook Report underscores the cost-effectiveness of proactive waste management strategies, which can significantly outweigh the economic and health costs of neglect. This realisation has spurred global initiatives to quantify and address the adverse outcomes associated with improper waste disposal, including pollution, health risks, and climate change implications.

In this context, the Department of Forestry, Fisheries and the Environment's (DFFE) collaboration with the German Development Cooperation on the Advanced Integrated Solid Waste Management (AISWM) Programme exemplifies a strategic approach tailored for South Africa. AISWM redefines integrated solid waste management (ISWM) through a blend of technological solutions and systemic reforms aimed at reducing landfill reliance (DFFE, 2015). This initiative emphasises the integration of efficient management systems with appropriate technologies to divert waste sustainably from landfills, marking a shift towards more holistic waste management practices.

Exploring the array of strategies for managing food waste in developing economies reveals a preference for biological and insect-mediated treatments over conventional landfilling. The DFFE's report highlights 12 alternative waste treatment technologies, emphasising the need to adapt these for food waste processing due to their varied applicability (DFFE, 2023).

Biological Treatment: Aerobic methods like windrow composting (WC) transform food waste into compost, enhancing soil fertility (Lu et al., 2020). Landfill gas capture (LFG), despite being a mainstream disposal method, faces challenges in biogas refinement (Nahman, 2011). Anaerobic digestion (AD) stands out for its versatility, converting a wide range of organic wastes into biogas, with the by-product serving as a high-quality fertiliser (Bernstad & la Cour Jansen, 2012). In-vessel composting (IVC) offers a faster, space-efficient alternative to WC suited for urban settings (Lu et al., 2020; DFFE, 2015).

Thermal Processing: Despite their potential for biofuel production, techniques like incineration, pyrolysis, and gasification are less effective with food waste due to their high moisture content. These methods are better suited for dry waste materials and will not be a focus of this study (Slorach et al., 2019).

Insect-Based Processing: Vermiculture and the use of Black Soldier Fly Larvae (BSF) exemplify innovative approaches to food waste conversion, producing valuable animal feed while reducing waste volume (Komakech et al., 2015). This method not only addresses waste management challenges but also contributes to the circular economy by creating high-value products from waste.

These technologies present promising solutions for food waste management in developing countries, offering pathways to sustainability and energy production. However, they also pose challenges related to operational complexity, technical requirements, and potential environmental impacts. A critical analysis of these technologies is essential to identify the most effective strategies for different contexts, ensuring that food waste management contributes positively to environmental sustainability and economic development.

2.3.1 Status quo: Landfilling

To understand the financial landscape of waste management accurately, it is crucial for municipalities to identify the comprehensive expenses tied to waste management functions. This task becomes challenging when costs associated with waste services are not distinctly allocated or catalogued in a financial compartment-specific to waste management (Godfrey & Oelofse, 2017). Municipalities and waste operators typically maintain detailed records of their assets, including their financial value and estimated residual lifespan, covering tangible and capital assets such as lands, infrastructure, and waste management tools (Godfrey & Oelofse, 2017). Budgetary components majorly include waste elimination, collection, sanitation services, and the mitigation or sealing off of existing landfills, alongside the budget for tool renewal or infrastructure enhancements.

2.3.1.1 Capital and operational costs

A review of local and international literature on various landfill management options is presented in Table 9, suggesting the cost implications for different landfilling methods. This literature indicates that smaller landfills bear a combined private cost of R280.00 per tonne of waste, while medium to large landfills have an average private cost of R507.00 per tonne of waste. The installation of gas collection and flaring systems is notably more expensive for smaller landfills, with costs becoming more manageable for larger landfills due to economies of scale. For this analysis, the cost of R564/t MSW for landfills with LFG capture is adopted, aligning with estimates by Leme et al. (2014) and other local studies indicating that landfilling with biogas capture and electricity recovery costs approximately R597.3/tonne MSW (BDA Group, 2009; Leme et al., 2014; Ferry et al., 2016).

Table 9: Summary of infrastructure costs for various landfilling options.

Type of landfill	Capital Cost (2022 ZAR/tonne)	O&M Cost (2022 ZAR/tonne)	2022 Total Cost (R/tonne)	Country	References
Dump	187.6	92.4	280	RSA	(DFFE, 2015)
Landfill without LFG collection	375.2 - 324.4	184.8 - 197.5	560 440 521	RSA RSA Australia	(DFFE, 2015) (De Wit and Nahman, 2009) (BDA Group, 2009)
Landfill with LFG collection flaring	367-564 +2.4(377.6)	197-282 +0.1(184.9)	564-846 (562.5)	Australia Brazil	(BDA Group, 2009) (Leme et al., 2014)
Landfill with LFG-E	417 412 +31(406.2)	- 185.3 +1.5(186.3)	602 597.3 592.5	RSA RSA Brazil	(DFFE, 2015) (Ferry et al., 2016) (Leme et al., 2014)

It's important to highlight that not all landfills are suitable for LFG capture projects. The viability of such projects requires a comprehensive feasibility study, with general guidelines suggesting a threshold based on the landfill's size and daily waste intake (Terraza & Willumsen, 2009). The South African guidelines for waste disposal by landfill indicate that the requirement for biogas capture is determined by a Site Investigation and Risk Assessment, with considerations for landfill gas migration and potential hazards or odour problems (DWAF, 1998).

Table 10 details the minimum landfill gas management requirements based on landfill size and leachate production, illustrating that only medium and large landfills typically require LFG capture and flaring to manage methane production in this report.

Table 10: Showing minimum landfill gas management requirements based on size and leachate production, either as required (R), not required (N) or where a feasibility study is required (F)

Landfill size	Communal landfill		Small landfill		Medium landfill		Large landfill	
Lining requirements	B-	B+	B-	B+	B-	B+	B-	B+
Gas management requirements	N	N	N	N	F	F	F	F
Gas drainage layer for capping	N	N	N	N	N	R	N	R

2.3.1.2 Potential Revenue: Gate Fees

Landfill entrance fees reflect the costs related to waste disposal, serving as a benchmark for evaluating advanced waste treatment alternatives. Disposal fees vary by waste category, with some landfills not charging for inert waste disposal, while waste from outside municipal regions might incur higher costs (DEFF, 2020). Cape

Town's landfill operations cost, for instance, was recorded at R640/tonne for the 2021/2022 fiscal year, which includes capital outlays, mainly for waste disposal and collection points (CoCT, 2021b). However, these figures do not account for land acquisition and infrastructure capital costs. Operational expenses of landfills are reported to vary between 150 to 450 ZAR/tonne, depending on management practices and tariff strategies (DFFE, 2015).

Table 11 showcases the 2021/22 municipal disposal gate fees for general and putrescible waste across various municipalities, highlighting the differences in disposal costs and indicating the contribution of each province to municipal solid waste generation in South Africa. This table, along with reports from Nahman (2011) and GreenCape (2022b), illustrates the variance in disposal fees, particularly noting Cape Town's high fees due to limited airspace and comprehensive cost accounting, contrasting with regions like the Northern Cape, where disposal charges are absent. The weighted average gate fee of R376.28 is thus not reflective of the full cost of landfilling, prompting a reliance on literature estimates for more accurate cost assessments.

Table 11: 2021/22 Municipal disposal gate fees for general waste and putrescible waste.

Municipality	2021/22 Tariff for safe disposal of putrescible (food) waste. certificates are issued (R/tonne)	Average safe disposal tariff (R/tonne)	2021/22 Tariff (R/tonne)	Average tariff (R/tonne)	Province	Contribution of each province to generation of municipal solid waste in South Africa in 2017 (%)
Ekhurleni	R 600.59	R 1,570.78	R 370.00	R 309.00	Gauteng	26
COJ	R 3,846.75		R 292.00			
Tshwane	R 265.00		R 265.00			
Ethwkwini	R 1,456.40	R 1,456.40	R 547.20	R 547.20	KwaZulu-Natal	18
Polokwane	R 133.37	R 133.37	R 60.69	R 60.69	Limpopo	8
Mbombela	R 2,004.28	R 2,004.28	R 200.00	R 200.00	Mpumalanga	4
N. Cape (All)	R 0.00	R 0.00	R 0.00	R 0.00	N. Cape	1
Rustenburg	R 328.40	R 328.40	R 123.15	R 123.15	Northwest	6
Buffalo City	R 463.33	R 439.64	R 399.00	R 298.16	Eastern Cape	11
Nelson Mandela Bay	R 415.94		R 197.31			
Mangaung	R 525.00	R 525.00	R 440.00	R 440.00	Freestate	11
Coty of CT	R 848.40	R 848.40	R 640.00	R 640.00	Western Cape	15
Weighted average of fees		**R 1,014.47**		**R 376.28**		**100%**

The UK's Landfill Tax exemplifies the fiscal approach to addressing landfill environmental externalities, encouraging investments in landfill diversion alternatives through financial penalties. This strategy has been adopted by several countries to enhance the attractiveness of alternatives to landfilling (DFFE, 2015).

2.3.1.3 External costs

The assessment of external costs of landfilling underscores a multi-faceted environmental and social challenge, encompassing greenhouse gas (GHG) emissions, local air pollutants (LAP), leachate, and disamenity costs. Yang et al. (2022) emphasises the dominant focus of existing literature on GHG emissions, noting a comparative scarcity of comprehensive studies on water and soil impacts. By employing emission factors and pollutant costs, such as those derived from the BDA Group (2009), this study aims to quantify these externalities in monetary terms.

The development of GHG emission factors for waste management practices in South Africa, as discussed by Friedrich & Trois (2013a), highlights the significant methane emissions from landfills, suggesting potential for energy recovery. The variability of landfill gas emissions across climate regions further complicates the equation, suggesting that a one-size-fits-all approach to waste management may not be suitable (Fourie & Morris, 2004). Schott et al. (2016) and Sauve & Van Acker (2020) point to the methodological and systemic differences in life cycle assessments of landfills, indicating a broad spectrum of global warming potential outcomes based on factors such as waste composition and landfill gas treatment technologies.

Table 12, by juxtaposing various GHG emission estimates from different landfill technologies and regions, underscores the variability inherent in these assessments. This variability is attributed to differences in waste composition, landfill operational efficiencies, and methodological assumptions, among others. The comparison suggests a general alignment of the BDA Group (2009) study's assumptions with those of Friedrich & Trois (2013a), particularly after adjusting for South African organic waste levels.

Table 12: Literature of the GHG emissions for various studies.

Landfill Technology	GHG Emissions (kg CO_2 eq/ton waste)	Region	References
Dumps	1,852.20	RSA	(Friedrich & Trois, 2013a)
Landfill without BG capture	2,127.09	RSA	(Friedrich & Trois, 2013a)
	1410-1530	Australia	(BDA Group, 2009)
	1286	Global	(Lou & Nair, 2009)
	1000	SS Africa	(Komakech et al., 2015)
	1550	Australia	(Lu et al., 2020)
Landfilling with gas collection and flaring	620.34	RSA	(Friedrich & Trois, 2013a)
	650-660	Australia	(BDA Group, 2009)
Landfilling with gas collection and electricity generation	531.72	RSA	(Friedrich & Trois, 2013a)
	500-550	Australia	(BDA Group, 2009)
	160	Meta-analysis	(Morrisa et al., 2013)

LAP estimates, drawing from the USEPA AP42 compilation, highlight the environmental and health risks of landfill emissions, with studies like Salami & Popoola (2023) emphasising the need for focused research on particulate matter impacts in low-income countries. The calculation of disamenity costs using the hedonic pricing method, which will be further explored in Chapter 5, aims to quantify the impact of landfills on property values, offering a nuanced understanding of environmental externalities.

The financial assessment by De Wit & Nahman (2009) places the cost of landfilling at R240 per tonne of waste, escalating to R351 per tonne for municipal solid waste in Cape Town. This estimate, however, does not fully account for the variability in climate, socio-economic factors, and operational standards across South Africa, potentially underestimating the external costs associated with food waste due to its higher organic content and associated nuisances.

2.3.2 Typical Revenue from AWT

AWT technologies present a spectrum of revenue-generating opportunities, influenced by the market demand for outputs such as compost, biogas, and biofuel. The variability in revenue streams is largely contingent upon local market conditions, quality standards, and the availability of feedstock. South Africa's demand for high-quality compost for organic farming, as noted by DFFE (2015), highlights the potential for revenue generation from composting operations.

The exploration of opportunity costs associated with the inedible portion of food waste by De Lange and Nahman (2015) sheds light on the potential financial benefits of diverting food waste from landfills to composting and bio-energy generation processes. Their analysis reveals the substantial cost, R5,963.00 per tonne, associated with landfilling food waste, emphasising the economic and environmental rationale for seeking alternative waste management strategies.

Biogas generation from food waste, despite the current small scale of operations in South Africa, presents a significant revenue potential when compared to the cost of liquefied petroleum gas. The composting potential, however, is limited by the need for optimal carbon-nitrogen ratios, highlighting the necessity of mixing food waste with other organic materials to produce high-quality compost.

2.3.3 Private Cost Benchmarks for AWT

The economic feasibility of AWT technologies is a critical aspect of their adoption, especially in developing contexts like South Africa. Benchmarking the costs associated with these technologies, including both capital and operational expenses, is essential for stakeholders to make informed decisions regarding the implementation of AWT solutions. This section delves into the cost implications of various AWT technologies, drawing upon an array of literature with a specific focus on data relevant to developing countries to offer a comparative perspective against first-world benchmarks.

The cost structure of AWT technologies varies significantly across different methods and geographical locations. Table 13 presents an extensive overview of expected cost ranges for various AWT options, with a particular emphasis on estimates from developing countries to provide a more relevant cost assessment for South African implementation. It is imperative to recognise that these figures should serve as a guide rather than definitive values due to the fluctuating nature of factors like labour wages, equipment costs, and the availability of maintenance services. The detailed breakdown in Table 13 encapsulates the capital and operational costs across technologies and countries, offering a broad spectrum of financial requirements for AWT deployment.

Table 13: Expected cost ranges for various AWT options

Technology	Capital Cost (2022 ZAR/tonne)	O&M Cost (2022 ZAR/tonne)	2022 Total Cost (ZAR/tonne)	Country	References
Windrow composting	168-224	252-336	420-560	RSA	(DFFE, 2015)
	-	-	153.8	Malaysia	(Yoshizaki et al., 2012)
	80-133	110-228	190-361	RSA	(GreenCape, 2021)
	-	-	195	China	(Liu et al., 2020)
	120	160	281	-	(van Haaren et al., 2010)
In-vessel composting	350-550	350-550	844-1,100	RSA	(DFFE, 2015)
	700-900	441	1141-1341	RSA	(GreenCape, 2021)
			5900	USA	(Mu et al., 2017)
			193	China	(Liu et al., 2020)
Anaerobic digestion	500-550	500-550	1,000-1,100	RSA	(DFFE, 2015)
	-	15.2	1,400	RSA	(Musvoto et al., 2018)
	-	-	179.5	Malaysia	(Yoshizaki et al., 2012)
	341	774	1115	RSA	(GreenCape, 2021)
	-	-	474-790	Pakistan	(Salam et al., 2021)
	119-650	9-65	128-715	Unknown	(Kumar et al., 2017)
Vermicompost	80-133	-	420-560	RSA	(GreenCape, 2021)
	-	-	121	Malaysia	(Lim et al., 2016)
	258	-	647	India	(Thirunavukkarasu et al., 2022)
Black Soldier Fly Processing	-	-	600-1,000	RSA	(GreenCape, 2023)
	-	-	500-650	RSA	(GreenCape, 2022a)
	-	-	94-253	Pakistan	(Salam et al., 2021)
	-	114-465	3253	UK	(Ffoulkes et al., 2021)
	NA	141	-	Indonesia	(Zurbrügg et al., 2018)
	84.3	531.6	615.9	Ghana	(Joly & Nikiema, 2019)
	61	543.5	604.5	Costa Rica	(Joly & Nikiema, 2019)
	NA	472.4	-	Indonesia	(Joly & Nikiema, 2019)
	341.6	-	NA	Canada	(Joly & Nikiema, 2019)
	146	-	NA	RSA	(Joly & Nikiema, 2019)

The data presented reveals significant variability in the costs associated with different AWT technologies. For instance, in-vessel composting and anaerobic digestion exhibit higher capital and operational costs, reflecting their complexity and the advanced technology required. Conversely, more traditional methods like windrow composting and vermicomposting show lower cost ranges, suggesting their suitability for contexts with limited financial resources. The emerging Black Soldier Fly (BSF) processing technology, despite its novelty and limited data availability, presents an intriguing cost profile that warrants further exploration, given its potential for waste valorisation.

The variation in costs also highlights the influence of local conditions on the economic viability of AWT methods. Factors such as the scale of the facility, local

wage levels, and the cost of equipment play pivotal roles in determining the overall financial requirements of implementing these technologies.

An analysis of the provided cost data allows for the derivation of average costs for each technology, offering a simplified yet insightful perspective on the financial considerations for AWT deployment in South Africa. The calculated averages serve as a benchmark for stakeholders, aiding in the assessment of tenders and proposals for AWT projects. It's noteworthy that these averages, while informative, must be contextualised within the specific conditions and requirements of the intended project location. WC - R308, IVC - R1,106.50, AD - R1,153.75 and VC - R437.00 per tonne of waste.

The case of BSF technology, despite its emerging status and sparse data, illustrates the potential for cost-effective waste treatment solutions. The compiled costs from various studies provide a preliminary financial outlook for this technology, emphasising the need for further research and development to fully ascertain its economic viability and scalability. The RSA capital cost is understood and reported as R146/tonne of waste. Taking an average of the reported O&M costs for Ghana, Costa Rica, and Indonesia (R515.8/tonne waste) gives a total cost of R661.8 per tonne of waste.

2.3.4 External costs and benefits

As with landfilling, the external costs associated with AWT are comprised of GHG emissions, LAP, leachate, and disamenity costs to the surrounding community. However, there are also potential benefits to the application of AWT that may offset these costs. All diversion options reduce the organic fraction sent to landfill, which directly impacts all externalities. Additionally, the application of by-products such as compost and digestate can sequester carbon in the soil and replace synthetic fertilisers (Morrisa et al., 2013). Moreover, the energy produced by these processes has the potential to offset energy generated by fossil fuels. A waste stream with a higher decay constant (DOC) will result in increased GHG emissions from aerobic decomposition, while waste with a lower decay constant requires a longer composting period, affecting the operational requirements for processing that waste stream (Lou & Nair, 2009).

A review of life cycle assessments on food waste management reveals significant variations in global warming potential (GWP), largely due to methodological differences and system boundary settings (Bernstad & la Cour Jansen, 2012; Schott et al., 2016; Morrisa et al., 2013). These differences in GWP are attributed not to actual environmental impact discrepancies but to variations in the performance of the studies (Bernstad & la Cour Jansen, 2012). Schott et al. (2016) highlight the influence of methodological choices and system boundary settings, leading to potentially misleading comparisons between different food waste management treatments. Particularly, assumptions about the waste management system's interaction with the background energy and/or bio-system are crucial to the outcomes (Schott et al., 2016).

LCA models are instrumental in evaluating the emissions from each AWT, with extensive meta-analyses like that by Morrisa et al. (2013) providing insights into the climate impacts of organic waste management methods. These LCAs suggest that aerobic composting and AD are generally preferable to waste-to-energy and landfilling with methane capture for energy generation in terms of just GWP. Morrisa et al. (2013) also noted that in terms of EP and ecotoxicity, aerobic composting ranks higher than AD, followed by LFG-E, with LFG collection and flaring being the least favourable across all categories. In terms of LAP, AD is ranked highest, followed by composting and LFG-E (Morrisa et al., 2013).

LCA data indicate that the environmental performance of AD is notably influenced by the quantity of methane produced for use as an energy offset, which varies based on the organic waste's composition and the specific AD technology employed. The benefit magnitude from energy offsets also depends on the type of displaced energy fuel, with larger climate benefits when displacing coal compared to natural gas. Notably, because AD aims to maximise methane production, any operational deficiencies leading to fugitive methane emissions can significantly affect this technology's environmental performance.

The environmental impacts of aerobic composting, including WC and IVC, are highly dependent on the management practices of compost facilities to maintain aerobic conditions. Deviations can lead to methane and/or nitrous oxide emissions, both

potent GHGs (Morrisa et al., 2013). Bernstad & la Cour Jansen's (2012) meta-analysis of 25 studies on food waste management technologies found that assumptions about energy substitution significantly impact results, with AD generally offering higher GWP benefits compared to composting across all scenarios.

Lim et al. (2016) explore the efficacy of composting and VC for nutrient recovery from organic waste, considering GHG emissions and economic viability. They suggest that intermittent aeration, bulking agents, and earthworm abundance can mitigate GHG emissions during these processes.

Lu et al. (2020) determine that centralised WC offers superior overall performance, with landfilling being the least favourable option from both environmental and economic perspectives. Energy consumption was identified as a major environmental impactor, more so than waste collection and transportation in centralised composting scenarios.

An LCA study by Fan et al. (2023b) shows that anaerobic technologies, including AD, exhibit better environmental performance in GWP, AP, EP, ADP, and ODP compared to aerobic technologies. The stages of pretreatment, biogas electricity generation, and composting are principal environmental impact sources, whereas the life cycle cost of AC is slightly lower than AD.

Komakech et al. (2015) found that an AD system has the best performance in terms of energy use, GWP, and eutrophication, followed by BSF and VC. However, a poorly managed AD system with significant methane losses can have poor GWP performance. The composting system analysed exhibited poor GWP results mainly due to direct emissions from the composting process. In terms of EP, AD performed the best, while BSF, VC, and WC had similar outcomes. Composting, despite being easy to implement and superior to landfilling in terms of GWP, could improve its performance with better management practices, making AD the most suitable option for SSA cities.

Slorach et al. (2019), using LCA, found AD to have the lowest impacts in 13 out of 19 ecological categories, including GWP, which is estimated to be net negative. Landfilling exhibited the highest GWP, while INV was generally the least favourable

option, highlighting AD's alignment with circular economy principles and environmental sustainability.

Kim et al. (2011) evaluated the costs associated with eight food waste disposal options, finding that diverting food waste to animal feed presented the best GWP, followed by WC and AD, with landfilling having the worst impacts. A study by Mertenat et al. (2019) suggests that BSF treatment emits 47 times less than WC.

This detailed literature review ranks AWT technologies for GWP as follows: AD significantly outperforms other technologies, followed by BSF, LFG-E, VC, WC, and IVC, with landfilling being the least favourable. The consensus indicates that anaerobic processes, particularly AD, surpass aerobic processes in terms of GHG emissions.

2.4 Critical evaluation of LCC and SCBA in food waste AWT technologies

The evaluation of advanced waste treatment (AWT) technologies for food waste necessitates a deep dive into both the environmental and economic realms. Yang et al. (2022) lay the groundwork by analysing various assessment methodologies, emphasising the prevalent use of environmental life-cycle costing (ELCC). This approach, however, lacks the ability to quantitatively compare environmental benefits against economic costs, underscoring a critical gap in the comprehensive assessment of AWT technologies. The integration of life-cycle assessment (LCA) with life-cycle costing (LCC) emerges as a pivotal strategy to bridge this gap, advocating for a more nuanced analysis that considers external cost monetisation and the refinement of environmental cost-effectiveness (ECE) metrics.

Diving deeper into the hierarchy of food waste management, it becomes evident that prevention surpasses treatment and utilisation, primarily due to the significant environmental and economic detriments averted by bypassing food production processes. However, when treatment is unavoidable, anaerobic digestion emerges as a more environmentally favourable option compared to composting, landfilling, and even biodiesel production (Mayer et al., 2020; Slorach et al., 2019; Martinez-Sanchez et al., 2016; Ahamed et al., 2016; Yu & Li, 2020). This preference, detailed

by Slorach et al. (2019), Edwards et al. (2018) and Kim et al. (2011), shifts when economic considerations are brought into the picture, with composting often emerging as a more cost-effective solution due to its lower operational and capital costs. This economic advantage of composting, however, does not diminish the value of anaerobic digestion, especially when considering its environmental benefits and potential for by-product revenue generation.

The incorporation of studies by Aye & Widjaya (2005) expands this analysis by comparing various composting options, AD, and LFG-E against open dumping in Indonesia. Their cost-benefit analysis, grounded in GHG savings, positions WC as having the highest benefit-cost ratio, despite AD's lower environmental impact. This discrepancy highlights the nuanced decision-making landscape, where the relative costs of technologies and the context of Indonesian energy production—marked by a scarcity of fossil fuels—play critical roles.

Similarly, Cabaraban et al. (2008) provide a comparative analysis of IVC and LFG-E in the USA, assessing both environmental impacts and economic costs. Their findings favour LFG-E over IVC for both financial and environmental benefits, aligning with broader trends that suggest the economic viability and sustainability of gas capture and energy generation from landfill gases.

The complex interplay of environmental benefits and economic costs is further complicated by the logistical considerations of collection and transportation, as highlighted by Eriksson et al. (2016). The efficiency of food waste treatment systems is significantly influenced by these operational aspects, with separate collection and transportation processes notably impacting the cost-effectiveness of anaerobic digestion compared to co-incineration strategies.

The discourse on the separation and treatment of food waste reveals a nuanced interplay between environmental impact and economic feasibility. Studies by Carlsson et al. (2015) and Yu & Li (2020) highlight the balance between separation rates and household time costs, suggesting a threshold where environmental benefits justify the economic costs. Moreover, the logistical aspects of collection and transportation significantly sway the environmental and economic evaluations of food

waste management systems, with Eriksson et al. (2016) shedding light on the competitive nature of central sorting versus source separation.

The SCBA of biodegradable waste management options by the 2002 Eunomia report and subsequent studies by Slorach et al. (2019) and Fan et al. (2023b) further enrich our understanding of these technologies' performance. These analyses reveal a preference for AD and its combinations with other technologies, not only for their superior environmental performance but also for their economic and social viability, particularly when product outputs are considered. Fan et al. (2023b) encourage the combination of BSF and WC with AD to further increase ecological, economic and social benefits; these combinations outperform AD and WC in isolation.

In synthesising these insights, it becomes clear that the choice of food waste AWT technologies is not merely a question of environmental versus economic benefit but a complex decision matrix that requires a holistic understanding of each technology's multifaceted impacts. As such, this critical evaluation underscores the need for a more integrated approach that balances these considerations, advocating for technologies that not only minimise environmental impact but also offer sustainable economic advantages.

CHAPTER 3: STATE OF WASTE IN SOUTH AFRICA: POLICY OVERVIEW

3.1 Reporting on State of Waste

In 2012, the Department of Environmental Affairs (DEA) issued its third national waste report, setting out to gather data on waste production, categorise waste types, and understand the disposal or treatment methods in use (DEA, 2012). This study offered a broad view of the national situation, but it underlined a notable deficit in precise waste data reporting, with a significant amount reported as 'unclassified waste' and failure to adhere to existing regulations.

The 2018 State of Waste (SoW) report analysed the factors fuelling waste production, future projections, and their implications on the growing volume and complexity of waste, and the pressure on the current infrastructure to manage these wastes (DEA, 2018). Similar to the 2012 report, the 2018 SoW underscored data availability as a significant hurdle, yet expressed a greater confidence level in waste generation estimates and classifications, evident in the marked reduction in 'unclassified waste' streams.

The DEA has released additional reports that are relevant to this literature review, including the 3rd South African Environment Outlook Report (DEA, 2018b), the National Pricing Strategy for Waste Management (DEA, 2016), and the 2nd South African Environment Outlook Report (DEA, 2016). Additional sources of reporting on the state of general waste landfills come from AfriForum's landfill site audit report. This report, with the support of the Department of Forestry, Fisheries and the Environment's (DFFE's) Waste Management Division, establishes the extent to which landfill sites (legal as well as illegal) in the municipalities of AfriForum's 160 branches across the country comply with the minimum requirements for compliance and to compare these with their waste management licences.

A waste research, development and innovation (RDI) capability mapping carried out by the Council for Scientific and Industrial Research (CSIR) and the Department of Science and Technology (DST) in 2014 identified the nascent yet promising research

capabilities within specific South African universities and science councils. They have contributed to a body of work that integrates academia-produced knowledge, showcasing South Africa's research into biomass and food waste valorisation as an alternative to landfill disposal (DST, 2014).

This groundwork serves as a cornerstone for cultivating research and development, human capital development, and innovation within the waste sector. However, a significant portion of this knowledge, primarily found in post-graduate dissertations, remains inaccessible to industry and government stakeholders. The DST is investigating strategies to unlock this information to foster evidence-based decision-making. This dissertation forms part of a series of planned academic publications aimed at bringing South Africa's research into the public sphere (DST, 2014).

3.2 Volume of Waste Generated

The 2012 third national baseline report found that in 2011, 90% of all waste generated in SA was sent to landfill, and 10% was recycled. (DEA, 2012) The 2018 SoW report is extensive and detailed, but it reported that in 2017, 20.6% of all waste generated was recycled, while 79.2% was landfilled. (DEA, 2018) The below table summarises the volume of waste generated in 2011 and 2017.

Table 14: Summary of generated waste volumes and the proportioned landfilled or recycled from both South African State of Waste Reports 2012 and 2018.

	2011			2017		
	Total Volume (Million Tonnes)	Proportion Recycled (%)	Proportion Landfilled (%)	Total Volume (Million Tonnes)	Proportion Recycled (%)	Proportion Landfilled (%)
Total Waste Generated	108	10%	90%	121.0	20.6%	79.2%
General Waste	59	10%	90%	54.2	38.6%	61.4%
Hazardous Waste	1	6%	94%	66.8	6.0%	93.7%
Unclassified waste	48	10%	90%	-	-	-

From 2011 to 2017, waste generation increased by 12.0%, according to data provided in these reports. It is important to note that the 2018 report paid specific attention to unclassified waste and categorised it as either general or hazardous.

South Africa has made significant strides in challenging waste management from predominantly landfilling in 2011 (90%) to recycling, with almost 40% of general waste in 2017 being recycled, as can be seen in Table 14 above. This shift can be attributed to recent changes in waste legislation, policies and plans, an increase in the use of waste as a resource, media awareness, and conscious consumerism (DEA, 2018) However, due to the increase in waste production over this period due to increased population, the volume of waste being sent to landfill (in tonnes) has not reduced significantly. There is still a significant dependence on landfilling as a waste management option, even though there has been a proportional decrease.

With landfill sites reaching capacity, especially in Gauteng and Western Cape provinces, it is becoming increasingly crucial to explore sustainable alternatives to waste management, including the conversion of organic waste into energy and value-added products (Creamer Media Reporter, 2018). The increased recycling rates are a positive development, but further efforts are needed to address the continuing reliance on landfill disposal and to capitalise on the potential economic and environmental benefits of waste conversion technologies.

Table 15: From the 2018 SoWR the percentage of municipal waste contribution by province in South Africa.

Province	Waste generated as a percentage of total waste
Western Cape	14.7
Eastern Cape	10.5
Northern Cape	1.1
Free State	10.5
KwaZulu Natal	17.9
Northwest	6.3
Gauteng	26.3
Mpumalanga	4.2
Limpopo	8.4

Further to 2012 and 2018 reports, the DFFE has initiated the South African Waste Information Centre (SAWIC). The South African Waste Information Centre (SAWIC) serves as a centralised repository for information and data related to waste

management in South Africa. Its primary purpose is to support the efficient planning and regulation of waste management services across the country. This involves the collection, analysis, and dissemination of data related to waste generation, recycling, treatment, and disposal. SAWIC aims to assist in the development of effective waste management policies, foster sustainable practices, ensure regulatory compliance, and promote awareness among stakeholders and the public about waste issues. Additionally, the centre is instrumental in tracking progress towards national waste management goals and in supporting the decision-making processes of government bodies, businesses, and other entities involved in waste management.

The below Table 16 table is a summary of data extracted from the SAWIC database indicating tonnage reports for each year from 2018. These tonnage reports would suggest that there is a significant decrease in waste going to landfill and even that the total volume of general waste produced is decreasing. From all projections discussed in the 2018 SoW report this should be increasing and not decreasing.

Table 16: Tonnage reports pulled from the SAWIC website indicating volumes of general waste processed from 2017 to 2022. (SAWIC,2023)

Year	Waste Disposed	Waste Treated	Waste recovered or recycled	Total Produced
2017	37,059,493.8	522,030.9	7,202,469.9	44,783,994.6
2018	26,502,667.8	6,914,002.0	16,890,647.3	50,307,317.1
2019	26,811,359.7	357,751.7	21,081,766.9	48,250,878.3
2020	21,205,676.5	905,277.9	3,264,054.3	25,375,008.7
2021	21,862,911.3	546,309.5	6,088,115.9	28,497,336.7
2022	19,331,359.5	652,118.7	8,587,591.9	28,571,070.1

The accuracy of the tonnage reports on the SAWIC depends on various factors, including the rigour of the reporting mechanisms, the compliance of the reporting entities, and the verification processes in place. The methodologies used to collect waste data can vary and may affect accuracy. Many municipal landfill sites do not have weighbridges and are often unmanned, without facilities to accurately record and report disposal volumes. The accuracy also hinges on how rigorously waste generators, handlers, and facilities comply with reporting requirements. Non-compliance or under-reporting by entities can lead to inaccuracies.

SAWIC is intended to provide a comprehensive and accurate reflection of waste statistics in South Africa, and efforts are made to ensure the quality of the data.

However, like any large-scale reporting system, there may be discrepancies and challenges that can affect the accuracy of the information provided. Due to these discrepancies, the audited and refined volumes in the 2018 SoW report will be used for analysis as they have been adjusted to account for these inconsistencies in reporting.

3.3 Classification of Waste Generated

The classification and categorisation of varying waste streams are of essential importance for solid waste management. For a comprehensive assessment of the state of waste management, a clear understanding of waste streams and how they are disposed of is required. The National Environmental Management: Waste Act of 2008 (referred to as the Act) clearly outlines the classification of waste streams, this then informs the minimum disposal requirements and local policy decisions. (NEMA, 2008) The efficacy of the Act requires an accurate baseline of waste information as evidence on which to base policy decisions and measure implementation. (DEA, 2012)

In 2011 cabinet approved The National Waste Management Strategy (DEA, 2011) which sets targets for the reduction of waste production, reuse, recycling and recovery. The strategy stipulates a 5-year monitoring period and hence accurate waste information is required. As a result of this legislation, the DEA implemented the South African Waste Information System (SAWIS), which requires waste producers to report on the waste streams, volumes, and treatment methods.

South African legislation broadly breaks waste into two categories: General Waste (GW) and Hazardous Waste (HW), based on the risk it poses. (DEA, 2018) The Act provides guidelines for the handling, storage, and disposal of each category of waste to ensure that they are managed in an environmentally responsible and sustainable manner. It is the responsibility of individuals, businesses, and governments to ensure that waste is managed in accordance with the Act.

The hazardous waste category, as defined by the Waste Amendment Act of 2014, includes waste that:

“contains organic or inorganic elements or compounds that may, owing to the inherent physical, chemical or toxicological characteristics of that waste, have a detrimental impact on health and the environment” (NEMA, 2014).

Hazardous waste can result from various industrial processes, medical facilities, and other sources that generate potentially harmful materials. Proper management and disposal of hazardous waste are crucial to minimise risks to public health and the environment.

In the 2018 SoW report, the classification of waste generated in South Africa was improved, with more accurate categorisation and a notable reduction in the volume of 'unclassified waste' streams (DEA, 2018). This improvement in waste classification contributes to a better understanding and management of waste streams, ultimately guiding the development of sustainable waste management strategies.

CHAPTER 4: METHODOLOGY

The methodology of this research is designed to rigorously assess the multifaceted issue of food waste management in South Africa, drawing upon a blend of quantitative and qualitative methods to ensure a comprehensive analysis. It is structured around three core methodological components, each chosen for its proven efficacy in environmental and economic analysis. This section introduces these components, explaining their relevance to the research aims and the broader context of sustainable waste management.

Firstly, this study leverages the most recent organic waste data provided by Oelofse et al. (2021), employing the Benefit Transfer Method (BTM) as adapted from the 2009 BDA Group report. This approach allows this study to estimate the full cost of landfilling in South Africa, including often-overlooked external environmental and social costs. By applying the BTM, initially utilised in Nahman et al. (2011) for Cape Town's landfills and expanding its application nationwide, this study adjusts for a variety of critical factors such as regional climate variations, landfill operational characteristics, and proximity to human settlements. This nuanced, localised analysis is pivotal for accurately assessing the cost landscape of landfilling across South Africa, setting a foundational understanding that is critical for the subsequent stages of this study.

Building upon the calculated landfilling costs, the second component of the methodology involves a thorough review of LCA data from existing literature pertaining to various organic waste conversion technologies. By synthesising this information, this report is positioned to apply a SCBA to each technology under consideration. This SCBA framework is instrumental in evaluating the total economic, environmental, and social costs and benefits of each technology, enabling the identification of the most viable solutions for reducing food waste, enhancing energy production, and creating value-added products within the South African context.

The third component of the methodology focuses on the development of policy and regulatory recommendations. Guided by the insights gained from the cost analyses

and SCBA, this stage involves a strategic assessment of how best to support and implement the identified waste conversion technologies. This includes considerations of policy adjustments, incentive structures, and regulatory frameworks necessary to facilitate the adoption and scaling of effective waste management practices.

4.1 Objective Definition

The objective of the analysis is to evaluate various organic waste management technologies, by obtaining total benefit (B_T), this is defined as the difference between revenue and costs, as seen in the below equation. The total benefit of each technology can then be compared considering private and external impacts. (Medina-Mijangos et al., 2020b)

$$B_T = B_P + B_E \quad (1)$$

Where B_P is the private benefit B_E is the external benefit.

4.2 Scope of the Study

This study seeks to analyse the effectiveness of selected organic waste conversion technologies in managing food waste in South Africa. The focus is to evaluate the potential of these technologies for landfill diversion, energy production, and the creation of value-added products. This study focuses only on the analysis of the treatment technologies, without considering the collection process, ranging from the arrival of the waste at the treatment facility to the sale to other intermediate agents in the value chain.

In doing so, the study aims to contribute to South Africa's commitment to achieving Sustainable Development Goal 12.3, which targets reduction in food waste and loss, by proposing strategies or technologies that enhance food waste management and ultimately reduce food loss and waste. For each technology a scenario was analysed to determine the feasibility of each option to treat food waste, there are defined as follows:

Scenario 1 (LF): Status quo of landfilling in South Africa.

Scenario 2 (LFG-capture): Landfilling with biogas capture and flaring (only applicable to medium and large landfills).

Scenario 3 (LFG-E): Landfilling with biogas capture and energy recovery (only applicable to medium and large landfills).

Scenario 4 (WC): Diversion of food waste from landfill with windrow composting (WC).

Scenario 5 (IVC): Diversion of food waste from landfill with in-vessel composting (IVC).

Scenario 6 (AD): Diversion of food waste from landfill with anaerobic digestion (AD).

Scenario 7 (VC): Diversion of food waste from landfill with vermicomposting (VC).

Scenario 8 (BSF): Diversion of food waste from landfill with black soldier fly processing (BSF).

4.3 Project Impacts

The effects of executing a municipal solid waste (MSW) management system, whether intentional or not, are measurable within a specific scope of study and can be categorised as either private or external (Medina-Mijangos et al., 2020b). Private, or internal, impacts encompass those directly tied to the MSW processing and subsequent resource recovery. These include the costs and benefits encountered by the entity financing or developing the project. On the expenditure side, private impacts consist of the capital (CAPEX) and operational (OPEX) expenses necessary to establish and maintain MSW management systems. Conversely, the positive aspects of private impacts involve the income from selling recyclable materials or energy produced by waste-to-energy plants. (Medina-Mijangos et al., 2020b)

Externalities refer to the unintended secondary impacts, which can be either beneficial or detrimental, stemming from an economic action (such as the operation of landfills) that do not affect the financial bottom line of the individuals or entities directly involved (like landfill owners or operators). Instead, these impacts are experienced by different segments of the society or may extend to impact future generations, spreading across the broader community. Such external effects linked

with the process of landfilling are typically not captured in the economic records of the landfill's proprietor or manager, yet they have broader implications on the welfare of the society (Nahman, 2011).

4.3.1 Impact identification

Using the inventory of impacts identified by Medina-Mijangos et al. (2020) and the Eunomia report prepared for the European Commission, the most relevant impacts can be identified for this study, for this study ensuring the total impact can be sufficiently covered.

Table 17: Summarising the impacts identified for this study.

Type of cost	Impact Group	Description of impact	Impact Identification	
			Cost	Revenue
Internal	Infrastructure	Treatment and final disposal	OpEx and CapEx	
Internal	Recovery of food waste	Gate Fees		Market Price
Internal		Compost Protein (feed) Electricity		Market Price
External	Use of materials	Landfill diversion		Avoid material sent to landfill
External	Environmental	Replace chemical fertiliser		Reduces nitrogen fert. use + CO_2 sequestration
External		Emissions to air	GHG production (BTM)	Reduction in landfill GHG Replace coal power
External		Emissions to air	LAP production (BTM)	Reduction in landfill LAP Replace coal power
External		Emissions to ground	Leachate (BTM)	Reduction in landfill leachate
External	Quality of life	Disamenities	HPS	

4.3.2 Impact Frequency

Impacts need to be located throughout the life of the project, and the period in which the impact occurs in the project lifecycle needs to be considered. A project can incur impacts at the beginning of the project (internal or external), during the operation of the project or after the close out. Landfills present a continued adverse impact even after closure. As such GHG emissions will be evaluated over the full life of the

landfill, or over the operational life of the evaluated technology and will be discounted accordingly. For homogeneity the analysis will be done per year of waste disposal.

4.3.3 Impact quantification

This report will identify the impacts on stakeholders in direct monetary terms. It is necessary to define the units carefully for each of these impacts studied as they will be the basis for economic valuations. It is proposed that the costs are referenced per year of the project lifecycle.

The BTM is used to estimate and translate the environmental and social impacts in 2022 ZAR per tonne of waste for the LF, LFG collection and LFG-E scenarios. It is important to note that this benefit transfer applies to the full mixed municipal solid waste (MSW) body and the annual quantity of generated waste will be used. The unit will therefore be 2022 ZAR per tonne of MSW.

For the analysis of each scenario, different technologies will process different annual volumes of waste, therefore, to work in homogeneous units that allow for the addition of homogeneous costs, the monetary costs need to be defined in 2022 ZAR per tonne of waste being processed.

4.3.4 Impact valuation

Some impacts can be quantified using monetary units. However, quantifying the external costs generally requires application of specific methods to quantify the benefits in monetary terms. The methods used in this study include BTM, HPS adapted from existing literature, and detailed literature reviews on LCA. Primary data requirements are listed below.

Private cost/benefit of landfilling and alternative technology:

Capital and operational costs are determined from a literature review, balanced against municipal estimates, this is further scaled per landfill as discussed in section 2.3.1.2 Potential revenue: Gate Fees. The income from landfilling is calculated from gate fees. Each technology provides an additional benefit from the value derived

from the diverted food waste, a market analysis was conducted for the current prices of these products.

External cost/benefit:

The external cost of landfilling was determined using the BTM adjusting studies done in Australia by the BDA Group for a South African context (BDA Group, 2009). The disamenity cost is taken from Nahman (2011) hedonistic pricing model which quantifies the cost of living near landfills in a South African context. This then allowed for the total external cost of landfilling in South Africa to be estimated. External costs for each technology were estimated from a detailed literature review.

4.4 Aggregation of costs and revenues

The compilation of both costs and revenues will facilitate a determination regarding the viability of investing in specific waste management technologies. This approach suggests quantifying costs and revenues in monetary terms per tonne of waste, tailoring these figures to the outputs associated with each technology under consideration. Here, revenues are interpreted as enhancements to human well-being, while costs represent detriments to human well-being. For a project or policy to be justified on a cost-benefit basis, its societal benefits must outweigh its societal costs.

The primary aim of this method is to assess the overall benefits of a project and to distinctly evaluate two scenarios: firstly, ensuring the MSW management system is operationally and financially sustainable, as indicated by a positive private benefit ($B_P>0$) – a scenario of particular interest to technicians and policymakers; and secondly, confirming that the MSW management system is viable from economic, financial, social, and environmental perspectives ($B_T>0$), a consideration of paramount importance to economists and the broader community. (Medina-Mijangos et al., 2020b)

4.4.1 Private Benefit

The private benefit (B_P) is obtained by subtracting private costs (PC) from private revenues (PR). Where the PR is the result of the sale price of recovered products (SP) per annual volume of waste treated or energy generated (AVW). The PC is made up of the sum of the investment cost (IC), operational and maintenance costs (OMC). The function is presented in the equation below:

$$B_P = \sum_{n=0}^{N}[(AVW_n \times SP) - (IC_n + OMC_n)] \quad (2)$$

Based on this equation, if $B_P > 0$, the project is operational economically and financially (Medina-Mijangos et al., 2020). For this analysis, the cost to finance the project and taxes are not considered and will need to be delt with on a project-by-project basis. Since the municipality owns the project, and are exempt from paying income tax, the tax is also not considered. Private companies would have to take this into account.

4.4.2 External Benefit

External benefits are described in the below equation as the sum of external revenue (PE), minus the sum of the external costs (NE).

$$B_E = \sum_{n=0}^{N}[(PE_n - NE_n)] \quad (3)$$

These externalities are obtained from both the positive and negative impacts from implementation of the chosen technology.

4.4.3 Total Benefit

The based on the above objective function to optimise is presented in:

$$B_T = \sum_{n=0}^{N}[(AVW_n \times SP) - (IC_n + OMC_n) + (PE_n - NE_n)] \quad (4)$$

CHAPTER 5: DETERMINING THE COST OF LANDFILLING IN SOUTH AFRICA

Previous studies have looked at individual landfills in South Africa to determine the cost of landfilling (Nahman, 2011) and the quantification of landfill externalities (Friedrich and Toris, 2013a, 2016). This report attempts to apply the same methodologies to 158 general waste landfills across South Africa. These landfills have significantly different climactic conditions as well as demographic variations which will all affect the cost of externalities and taking each unique site will make this analysis prohibitive. Some assumptions will need to be made to simplify the analysis and get a good estimate across South Africa.

To determine the costs of landfilling in SA the following is required:

1. Private costs and benefits – reviewed in Chapter 6
2. Quantification of the landfill environmental externalities in 2022 ZAR
3. Quantification of the landfill social externalities in 2022 ZAR

This report will determine environmental externalities using the benefit transfer method and the social disamenity cost from a literature review of previous work using the hedonistic pricing model both in South Africa and internationally.

5.1 Benefit Transfer Method

Understanding the full extent of the economic impact of landfill waste disposal requires an evaluation that goes beyond direct financial expenses. This assessment must incorporate the environmental, health, and social costs, which are frequently not accounted for in market transactions—collectively referred to as externalities (Nahman, 2011). These encompass the release of methane and other greenhouse gases, the risk of soil contamination from harmful leachates, and a range of additional adverse effects (BDA Group, 2009). To quantify these environmental costs, the benefit transfer method comes into play, using economic values derived from existing studies and adjusting them to fit the specific circumstances of the current landfill scenario. This adjustment process considers variations in the landfill's

design, waste composition, operational attributes, and local climatic conditions that affect leachate and landfill gas (LFG) generation and control (Nahman, 2011).

5.1.1 Methodology

This methodology extends to the impact pathway—how the disposed waste translates into economic costs or savings. It scrutinises variables such as the nature of the waste, landfill operational practices, and the regional climate, which together influence the volume and effects of LFG and leachate (Nahman, 2011; Friedrich and Toris, 2013a, 2016). The research also incorporates the environmental benefits derived from the displacement of pollution through the energy recovery from LFG, including details about the energy conversion systems involved and the type of energy production that is supplanted by this process (BDA Group, 2009; Friedrich & Toris, 2016).

By drawing on existing evaluation estimates done by the BDA Group (2009), in an Australian context, the calculation of unit costs for leachate and local air pollutants (LAP) is fine-tuned by factoring in geographical and socio-economic conditions. This adjustment of the 2009 BDA study allows the impact estimates to be applied in a South African context.

Several elements influence these cost relationships. For pollutants tied to LFG, air pollutants and leachate, the determinants include:

Waste categorisation and content, especially regarding organic waste fraction, since the degradation of biodegradable organic waste significantly dictates the production and composition of LFG and leachate (Nahman, 2011).

The design and operational features of the landfill, specifically whether there are systems in place to manage, treat, or control LFG and leachate emissions (Eshet et al., 2006). Modern landfills often minimise environmental harm through technologies that collect, contain, or treat emissions. For instance, advanced landfills with liners or where leachate is processed to an acceptable standard are less likely to contaminate the environment significantly (Eshet et al., 2006). Moreover, the capture and utilisation of LFG to generate energy (either as electricity or heat) significantly

mitigate methane emissions (European Commission, 2000). Older landfills lacking these mitigative systems typically result in higher levels of emissions (European Commission, 2000b). Climatic conditions impact LFG emissions, while leachate production is influenced by the local water balance, which depends on precipitation, evaporation rates, and soil types (Nahman, 2011).

The energy recovery benefits related to pollution displacement are influenced by:

The efficiency and design of the LFG capture system and the type of energy conversion (efficiency of LFG collection; methane concentration; energy content; recovered energy form; conversion efficiency), which determines the level of pollution offset (BDA Group, 2009). The type and efficiency of the energy generation that the recovered energy replaces. For instance, if the recovered methane substitutes power from an inefficient coal-fired station, the pollution displacement benefits are more substantial than if it were replacing a cleaner energy source (Nahman, 2011).

Similarly, for both leachate and local air pollutants, unit costs are swayed by:

The physical location and characteristics of the landfill site (such as its proximity to populations, topography, and meteorological conditions like wind patterns) (Eshet et al., 2006). The socioeconomic profile of the surrounding population, including population density and their willingness to pay for environmental quality, is influenced by both income levels and environmental preferences. However, when it comes to global warming impacts from CO_2 and CH_4 emissions, these are uniformly costed on a global scale because the resultant climate impacts are independent of the emission source (BDA Group, 2009).

5.1.2 Waste Types and Composition

To equate the Australian data to a South African context this report needs to compare the organic waste fraction in the landfills. Variations in the organic waste fraction impacts the leachate and LFG production and therefore the resulting impacts of these. The Australian Department of Climate Change, Energy, the Environment and Water, in their 2022 National waste report, estimated that the fraction of

organics disposed to landfill in 2020-21 was 47% (Ministry for the Environment, 2009).

South African waste data suggests the organic proportion of organic waste in general waste landfills is approximately 27%, as discussed previously in 2.1.3 Estimate of Food Waste Fraction Landfilled. Similarly, Nahman (2011) suggested that landfills in SA have a lower organic fraction, which will require an adjustment factor. The extent of the adjustment, based on the above organic portions, will be less than that used in Nahman 2011 based on the latest available waste data. The estimates from the Australian study will be multiplied by a factor of 0.57 as per the below equation.

$$Adjustmant\ factor = \frac{\%\ Organic\ Fraction\ SA}{\%\ Organic\ Fraction\ Australia} = \frac{27\%}{47\%} = 0.57$$

5.1.3 LFG and leachate emissions

It is typical for studies on landfill externalities to report emissions as unit costs per kg of emission (Nahman, 2011). This enables accurate linkages between emissions and the cost to society, as long as the emissions can be measured or accurately calculated the external cost can be determined (Mijangos et al., 2020). However, for sites where the emission data is not monitored or there is insufficient data, the cost per kg of emission cannot be calculated. Since there is limited data on the 158 sites being analysed it is more feasible to use the unit cost per tonne of waste (Nahman, 2011).

5.1.3.1 Climate

A contributing reason why the BDA Group (2009) study was selected for the benefit transfer is the correlation between Australian climate categories and climate categories in South Africa. This greatly simplifies the benefit transfer as no adjustments need to be made for climatic conditions.

5.1.3.2 Factors affecting pollution displacement benefits

In their analysis, BDA Group (2009) makes certain assumptions regarding landfill gas (LFG) collection and energy conversion. They estimate a 60% efficiency in LFG collection, attribute an energy content of 55.52 MJ/kg to methane, utilise a conversion factor of 3.6 MJ/KWh, and assume a 30% efficiency in converting methane to electricity. However, given that energy recovery systems are still developing in South Africa, it remains unclear whether these assumptions are entirely relevant to the South African context. Therefore, this study will not modify these estimates to account for potential differences in LFG capture and energy conversion technology specific to South Africa.

Additionally, the assumption by BDA Group (2009) that the energy generated would replace that from coal-fired power stations holds true for South Africa, where the majority of energy production is coal-based. Consequently, no alterations will be made to the assumptions regarding factors influencing pollution displacement benefits in the South African context. (Nahman, 2011)

5.1.3.3 Site location and Physical Characteristics

Considering that the impacts of CO_2 and CH_4 on global climate are not contingent on their emission source, there is no necessity to modify greenhouse gas estimates based on proximity to populated areas or specific physical features of the site (European Commission, 2000). Conversely, the influence of LAP is influenced by factors such as the proximity to human populations, local topography, and prevailing meteorological conditions, like wind direction (Eshet et al., 2006). Similarly, the implications of leachate emissions are more site-specific and should be adjusted to reflect each site's unique location and physical attributes (European Commission, 2000).

In this study, which examines 158 landfill sites across South Africa, there is a marked variation in terms of their location and physical properties. Therefore, adjusting for these site-specific differences presents a significant challenge. This complexity is mirrored in the BDA Group (2009) study, which encompasses a diverse range of landfill sites in Australia, each with its distinct set of location and physical

characteristics. As a result, similar to the conclusions drawn in Nahman's (2011) study of Cape Town landfills, this study also could not feasibly adjust for site location and physical characteristics in its analysis.

Incorporating population density into this analysis presents a notable challenge, given that the original study encompasses a wide array of landfill sites across Australia. While the study does offer distinct estimates for local air pollution (LAP) in urban versus rural locations, it lacks specificity regarding the particular sites involved or their respective population densities. Consequently, adapting the analysis to account for variations in population density between the sites examined in the original Australian study and the 158 landfill sites under consideration in this study proved unfeasible. Therefore, this study proceeds under the implicit assumption that the population densities surrounding urban landfill sites are comparably similar between the original Australian context and the South African context, and similarly for rural sites.

5.1.4 Socio-economics characteristics

In analysing the impact of greenhouse gases, it is essential to note that their climate impacts are uniform, irrespective of the emissions' origin. Therefore, in Table 18, the estimates for greenhouse gases do not necessitate adjustments based on site-specific socio-economic characteristics. However, when considering local air pollution (LAP), leachate emissions, and disamenities, these factors require more nuanced consideration. Monetary valuations for these are dependent on the size of the population exposed, which relates to the population density near the landfill. Additionally, willingness to pay is influenced by income levels (which dictate the ability to pay) and the preferences for environmental quality. These factors can significantly vary between countries, especially when comparing developed and developing nations (European Commission, 2000).

To adjust for differences in income and environmental preferences, the following formula is used (European Commission, 2000; Nahman, 2011):

$$\overline{E_{SA}} = E_{AUS} \times \left(\frac{I_{SA}}{I_{AUS}}\right)^{Q}$$

Where:

E_{SA}= The adjusted estimate (applicable to South Africa)

E_{AUS}=The original estimate (from Australia)

I =Per capita income at purchasing power parity (PPP) rates

Q = Income elasticity of demand for environmental quality

To account for income variances between countries, it is crucial to consider both the income levels and purchasing power parity (PPP), which reflects the actual buying power of income in different countries. For this study, per capita gross national income (GNI) at PPP rates for 2008 (the relevant year for the Australian data) were sourced from the World Bank (2011). The figures are as follows:

I_{SA}= 12,658.00 (current international dollars for 2008)

I_{AUS}= 37,533.00

(World Bank, 2011)

The income elasticity of demand for environmental quality is a measure of how demand for environmental quality changes in response to income fluctuations. Generally, it is theorised that environmental quality is perceived as a 'luxury' good, meaning demand increases with rising income, implying Q>1 (European Commission, 2000). However, empirical studies suggest a range of values, with many indicating income elasticities around 0.3 (European Commission, 2000a). Due to these varied findings, this study assumes Q=1.

Thus, to account for socioeconomic differences, the Australian estimates for local air emissions and leachate need only to be multiplied by the ratio of ISA/IAUS (which equals approximately 0.34). This adjustment reflects the socioeconomic disparities between the two countries.

5.1.5 Synthesis: Adjustments

In light of the factors outlined in the preceding sections, the calculations from the BDA Group's 2009 study have been modified as presented in Table 18. It's essential to factor in differences in waste composition, which require a 0.57 adjustment factor for all estimates. Additionally, for local air pollutants and leachate, a further

adjustment factor of 0.34 is necessary to accommodate socio-economic disparities. Consequently, the adjusted figures for greenhouse gases are obtained by multiplying the original estimates by 0.57. In contrast, the values for local air pollutants and leachate are adjusted by a factor of 0.19, derived from multiplying 0.57 by 0.34. While the consideration of disamenities is included and will be discussed in subsequent sections, it should be noted that these will not be factored into the later analysis.

Table 18: Range of monetary costs per tonne of waste for different emissions for three climate regions (adjusted from BDA Group, 2009)

Pollutant	**Greenhouse gas**				**Local air pollutants - urban**			**Local air pollutants - rural**			**Leachate**			**Disamenity**	
Assumptions	No LFG Capture	LFG Capture	LFG capture with ER	ER over operating life & post closure	No LFG Capture	LFG Capture	LFG capture with ER	No LFG Capture	LFG Capture	LFG capture with ER	No Liner	Liner	Best Practice	Non-compliant - Urban	Non-compliant - Rural
Climate: wet temperate region															
2008 A$	12.00	5.00	0.00	-1.00	0.68	0.55	0.97	0.21	0.24	0.09	0.02	0.01	1.00	10.00	5.00
2008 Rands (unadjusted)	83.16	34.65	0.00	-6.93	4.71	3.81	6.72	1.46	1.66	0.62	0.14	0.07	6.93	69.30	34.65
Adjustment Factor	0.57	0.57	0.57	0.57	0.19	0.19	0.19	0.19	0.19	0.19	0.19	0.19	0.19	0.19	0.19
2008 Rands (adjusted)	47.40	19.75	0.00	-3.95	0.91	0.74	1.30	0.28	0.32	0.12	0.03	0.01	1.34	13.43	6.72
2022 Rands	96.47	40.20	0.00	-8.04	1.86	1.50	2.65	0.57	0.66	0.25	0.05	0.03	2.73	27.33	13.67
Climate: Dry Temperate Region															
2008 A$	9.00	5.00	0.00	-1.00	0.67	0.54	0.96	0.21	0.23	0.08	0.01	0.00	1.00	10.00	5.00
2008 Rands (unadjusted)	62.37	34.65	0.00	-6.93	4.64	3.74	6.65	1.46	1.59	0.55	0.07	0.00	6.93	69.30	34.65
Adjustment Factor	0.57	0.57	0.57	0.57	0.19	0.19	0.19	0.19	0.19	0.19	0.19	0.19	0.19	0.19	0.19
2008 Rands (adjusted)	35.55	19.75	0.00	-3.95	0.90	0.73	1.29	0.28	0.31	0.11	0.01	0.00	1.34	13.43	6.72
2022 Rands	72.35	40.20	0.00	-8.04	1.83	1.48	2.62	0.57	0.63	0.22	0.03	0.00	2.73	27.33	13.67
Climate: Wet Tropical Region															
2008 A$	13.00	5.00	-1.00	-1.00	0.66	0.54	0.96	0.20	0.23	0.08	0.03	0.01	1.00	10.00	5.00
2008 Rands (unadjusted)	90.09	34.65	-6.93	-6.93	4.57	3.74	6.65	1.39	1.59	0.55	0.21	0.07	6.93	69.30	34.65
Adjustment Factor	0.57	0.57	0.57	0.57	0.19	0.19	0.19	0.19	0.19	0.19	0.19	0.19	0.19	0.19	0.19
2008 Rands (adjusted)	51.35	19.75	-3.95	-3.95	0.89	0.73	1.29	0.27	0.31	0.11	0.04	0.01	1.34	13.43	6.72
2022 Rands	104.51	40.20	-8.04	-8.04	1.80	1.48	2.62	0.55	0.63	0.22	0.08	0.03	2.73	27.33	13.67

5.2 Disamenities: Hedonistic Pricing model

The research includes an examination of social costs, particularly the disamenities experienced by residents residing in proximity to landfills. These fixed externalities, distinct from variable externalities influenced by waste volumes, persist irrespective of the amount of waste processed, primarily due to the landfill's mere existence (Du Preez & Lottering, 2009). A range of methods are employed to estimate these disamenity costs, including the hedonic pricing method (HPM), as utilised in studies by du Preez and Lottering (2009) and Nahman (2011).

In the context of South Africa, du Preez and Lottering (2009) explored the correlation between a property's proximity to a landfill site and its market value in New Brighton township within the Nelson Mandela Bay Metropolis, Eastern Cape. Their findings indicated a negative correlation, where property values increased with every 100 meters further from the landfill, leading to a cumulative regional disamenity cost of R1.4 million (Du Preez & Lottering, 2009). With a reference to the disposal rate of 20,000 tonnes of MSW per day, the disamenity cost was estimated around R5.83 per tonne of MSW.

Complementing these findings, Nahman (2011) conducted a similar study in Cape Town, employing HPM to demonstrate a decrease in house prices in closer proximity to landfills. Nahman (2011) assessed the disamenity cost per tonne of MSW entering landfill sites in Cape Town at approximately R57.46 (2010 prices), accounting for nuisances such as odours and pests.

Internationally, Australian studies by the BDA Group (2009) reveal that disamenity costs vary based on population density and landfill management practices, indicating that better-managed sites in urban areas range from 1 to 10 AUD per tonne of MSW, while in rural areas, the cost ranges from 0.5 to 5 AUD per tonne.

The European Commission (2000) suggests a broader estimate of 8 Euros per tonne of MSW, drawing on HPM studies conducted in the United States, albeit with acknowledged limitations in accuracy. Additionally, a study by Eunomia (2002) for the EU estimated disamenity costs between 1.27 and 2.75 Euros per tonne for Spain

and Greece. These countries were highlighted due to their closer alignment with South Africa's purchasing power parity and waste management practices, characterised by limited source separation and biodegradable waste diversion from landfills. Both Spain and Greece, having lower population densities around their landfills, reflect a reduced disamenity cost (Eunomia, 2002).

The below Table 19 summarises the literature on disamenity costs. In order to have comparative values these need to be adjusted for PPP as well as waste composition. Since there is limited data on the proximity to populations this will not be adjusted for, but the upper and lower limits will be applied to rural or urban areas under the assumption that higher population densities will be worse affected.

Table 19: Summary of literature on disamenity costs using HPM.

Landfill	Location	Disamenity Cost	Population density	Reference
Ibhayi Landfill	New Brighton, NMMM, Eastern Cape	R1.4 million R36.00 (0.44%) per 100m R5.83/t MSW	Moderate density	(Du Preez and Lottering, 2009)
Belville South	Cape Town, Western Cape	R566 million R57.46/t MSW	High Density 59,284 in 4km	(Nahman, 2011)
Coastal Park	Cape Town, Western Cape	R871 million R57.46/t MSW	Moderate density 30,821 in 4 km	(Nahman, 2011)
Vissershok	Cape Town, Western Cape	R0.00 /t MSW	Rural	(Nahman, 2011)
-	Australia	1 AU$/ton MSW 0.5 AU$/ton MSW 10 AU$/ton MSW 5 AU$/ton MSW	Best prac. – urban Best prac. – rural Non-comp – urban Non-comp - rural	(BDA Group, 2009)
-	US	8 euro/ton		(European Commission, 2000)
-	Greece Spain	1.38 Eu/t-2.75 E/t 1.27e/t-2.54e/t	Low density Low density	(Eunomia, 2002)

The adjustment calculations are summarised in Table 20, where the same methodology applied in section 5.1.4 and 5.1.2 was used for adjusting for PPP and organic waste fractions. The PPP for each country was taken from the World Bank database for each year in question (World Bank, 2011), the biodegradable waste fraction was taken from the Eunomia (2002) report as well as the BDA Group (2009) report.

Table 20: Adjustment to literature on disamenity costs by PPP and organic waste fraction.

Country (Location)	Disamenity cost at time of study (currency /tonne MSW)	Disamenity in 2022 ZAR (R/tonne MSW)	Organic waste fraction of MSW (%)	PPP and organic adjustment factor ($I_{SA}/I_{Country}$)	Equivalent adjusted disamenity cost in 2022 ZAR (R/tonne MSW)	Reference
South Africa (Urban)	5.38 ZAR	11.86	-	1	11.86	(Du Preez and Lottering, 2009)
South Africa (Urban)	57.46 ZAR	105.23	-	1	105.23	(Nahman, 2011)
South Africa (Rural)	0.00 ZAR	0.00	-	1	0.00	(Nahman, 2011)
Australia (Rural)	0.50 – 5.00 AUS	7.00-70.13	47	0.20	1.40-14.00	(BDA Group, 2009)
Australia (Urban)	1 – 10 AUS	14.03-140.25	47	0.20	2.80-28.05	(BDA Group, 2009)
Europe (NA)	8 EUR	166.75	40	0.25	41.69	(European Commission, 2000)
Greece (Rural)	1.38 – 2.75 EUR	32.84-65.45	49	0.22	7.22-14.40	(Eunomia, 2002)
Spain (Rural)	1.27 – 2.54 EUR	30.23-60.46	44	0.23	6.95-13.91	(Eunomia, 2002)

From Table 20, the relative disamenity costs between studies vary considerably. To decide on estimates for disamenities in South Africa, this report would propose dividing the disamenity costs between rural and urban areas as well as best practice and non-compliant landfills as done in the BDA Group (2009) report. Base on Table 20 this report proposes a similar scale used in the BDA Group (2009) report, 1:10 for best practice and non-complaince using the highest cost for urban non-compliance costs, this it then scaled across each category as shown in Table 21.

Table 21: External cost of disamenities proposed for this study.

Best Practice – Urban (R/tonne MSW)	Best Practice – Rural (R/tonne MSW)	Non-Compliant – Urban (R/tonne MSW)	Non-Compliant – Urban (R/tonne MSW)
10.52	5.26	105.23	52.62

The assumptions made in Table 21 align with the literature on disamenity costs with the proposed figures matching the range of values in the literature.

5.3 Pricing landfill externalities across South Africa

In order for the application of the calculated impact externalities to be applied to the 158 landfills across South Africa, each of the landfills is categorised by; landfill size, water classification which talks to its leachate production and lining requirements, climate region, whether the site has LFG systems and the kind of systems, impact zone (rural or urban), compliance with minimum requirements. With this the valuation of the external impacts for each landfill can be calculated.

The 158 landfills reviewed nationwide reflect only active landfills and does not include any landfills that have been decommissioned. The below sections review the assumptions and adjustments made to categorise and analyse the sites.

5.3.1 Review of externalities

The external costs of landfilling general waste are estimated to range from R3.96 to R211.62 per tonne of MSW in urban areas and -R2.56 to R157.84 per tonne of MSW in rural areas.

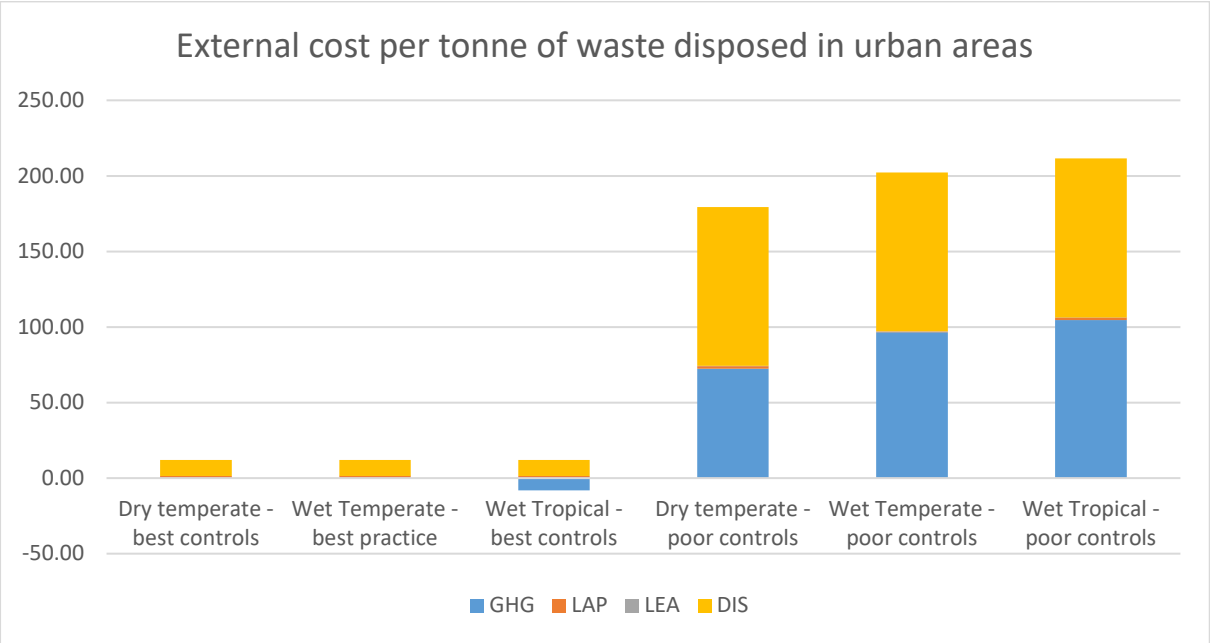

Figure 4: Graph showing external costs per tonne of MSW disposed in urban areas per climate region and compliance.

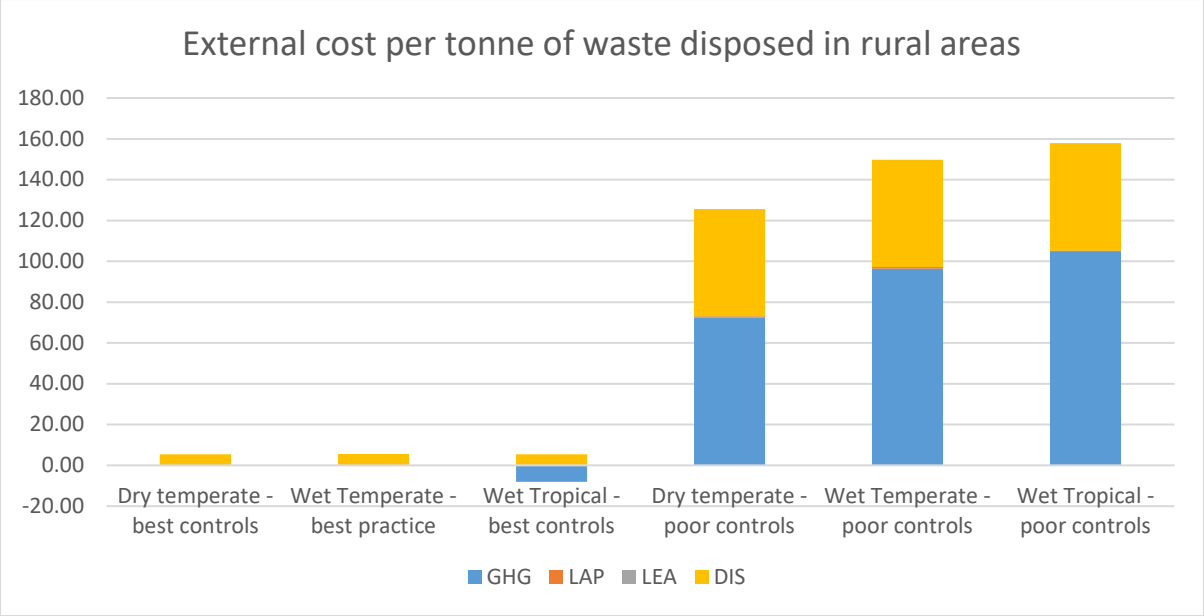

Figure 5: Graph showing external costs per tonne of MSW disposed in rural areas per climate region and compliance.

For urban areas with poor landfill controls, the GHG and disamenity cost make up the bulk of the external costs. For urban landfills with liners, LFG collection, energy recovery and best practice management, the disamenity and LAP are the greatest impacts as opposed to GHG emissions. Rural areas with best practice controls are mainly categorised by the disamenity cost.

These estimates are in line with estimates from the DBA Group (2009) study and European Commission (2000) studies. A sensitivity analysis will further determine the impact of each assumption.

5.3.2 Transport externalities

Transport externalities from waste collection and transport to each of the 158 landfills are not considered due to the complexity and lack of transportation data for each landfill. Further it is argued that these should be excluded, since alternatives to landfilling will also require transport. Further, it should be borne in mind that the objective here is to estimate the externalities specifically associated with the disposal of food waste to landfill to compare to alternative disposal options (De Lange & Nahman, 2015).

5.3.3 Landfill size

Each landfill being reviewed was given a size based on the classification as per the waste use licence issued by the DFFE. The size of the landfill is of particular importance to estimate the private cost of the landfill as well as feasibility (cost per tonne of waste) of expensive LFG management systems. The cost to install LFG collection systems in small and communal landfills is prohibitive, and these sites will be excluded and assumed to remain as they are.

It is important to note that while landfill size is important for the cost-benefit analysis, it does not impact the benefit transfer from the BDA report. Table 22 summarises the criteria used to size landfills by waste received per day. The external costs are calculated per landfill in ZAR per ton of MSW, to get an average external cost for the province and country these should be multiplied by the volume of waste received to scale the cost accordingly. However, due to the lack of site-specific data, another method is to weigh each landfill by size (estimated MSW received) and calculate the weighted average of the external cost.

Table 22: Size classification as per DFFE waste use licence

Landfill size classification	Min-Max rate of deposition (tonnes per day)	Average assumed deposition (tonnes per day)	Weighting
Communal (C)	0-25	13	1
Small (S)	25-150	88	7
Medium (M)	150-500	325	26
Large (L)	500-upwards	750	60

5.3.4 Climate

Each province was allocated a climate group using relevant weather data from the South African Weather Services historical database (South African Weather Services, 2022). Using the classification scheme described in the IPCC 2006 guidelines, which was also used in the BDA Group (2009) study, each province was allocated a climate region (IPCC, 2006). Table 23 summarises the findings. Each landfill will have an external cost depending on its climate category of the province it is located in, these external costs are found in Table 18. While there is significant

variation of climate over each province in SA, for simplicity of calculations, climate has been averaged over each province.

Table 23: Climate classification for each province in SA using IPCC 2006 climate categories (IPCC, 2006).

Classification	RSA Province	Average Normal day temp (°C) (1991-2020)	Average annual normal rainfall (mm) (1991-2020)
Wet temperate	GAU, EC, FS, MPA	10<AVG<18	>500
Dry temperate	NC, NW, WC	10<AVG<18	<500
Wet Tropical	KZN, LIM	>18	>500

(South African Weather Services, 2022)

5.3.5 LFG collection and leachate management systems

In South Africa, there exists a range of adherence levels regarding the implementation of engineered liners or leachate detection layers in landfills. This study aligns with the approach adopted by Nahman in 2011, focusing on marginal costs associated with waste management — specifically, the cost incurred for each additional tonne of waste deposited in landfills. The primary goal here is to furnish relevant data for evaluating current and future waste management strategies. It is important to note that this analysis does not delve into historical issues related to leachate resulting from waste previously deposited in landfills. The scope of this report is forward-looking and does not address or influence past landfill practices (De Lange & Nahman, 2015).

Due to the lack of specific leachate containment compliance on each of the 158 landfills nationwide it is assumed that landfills comply with their respective licences in terms of lining requirements, which has a direct impact on leachate. Assuming licences with B+ ratings are lined and compliant. Any ratings of B- are assumed to be unlined. This may not strictly be the case, as many B+ rated landfills have degraded leachate detection, collection, and containment systems. Auditing these systems is difficult to determine as the lining is buried in the waste body.

With respect to LFG collection systems, flaring or energy production a number of landfills were identified as having these systems installed as discussed in section 2.2.5 Waste-to-Energy, Table 7 summarises all these active LFG projects in South Africa. As this is an analysis of the status quo of the cost to landfill in SA the

associated costs from the BDA Group (2009) report will be attributed to those landfills only. Not all the sites listed in Table 7 are accounted for as some have been decommissioned and this study is concerned with active landfills. Of the nineteen municipal waste-to-energy programmes, five are operational LFG-to-electricity programmes, of these only two are currently receiving waste (both located in Gauteng). The three that are no longer receiving waste cannot benefit from all the technologies being analysed and therefore will not be considered, all are in KZN.

A further ten have LFG collection systems only, the seven in the Gauteng region are operational and are planning future LFG-to-electricity installations. Of the three in the Western Cape, two are currently receiving waste and the third closed in 2023 (not considered). The remaining 4 are in early planning stages.

The reduction in externalities has been attributed only to the 9 operation landfill sites.

5.3.6 Site location and physical characteristics

The BDA Group (2009) study covers a range of different landfill sites across Australia and classifies them as either being in urban or rural areas. The study is not explicit regarding exactly which sites are included, or their respective population densities. As such, it is assumed that population densities around urban sites are similar between the two studies likewise for rural sites. The challenge in this study is to classify all 158 landfill sites as either being in a rural or urban area. The following approach was used:

1. A GIS model (Using QGIS 3.32) was used to map all the landfill sites in South Africa, this was challenging, and each had to be found manually as not all have registered address. Google satellite image was the first layer on the model which helped identify landfills and their proximity to infrastructure.
2. South African National Landcover Data was downloaded from the DFFE database, this GIS data was used to indicate the location of the landfills and their proximity to rural and urban centres.
3. A further data set was imported which gave a rough indication of population density, this was overlayed on the above two layers.

4. Each landfill was then categorised as being rural or urban due to proximity to infrastructure or indicative population concentrations within a 3-5km radius of the site.

Figure 6, Figure 8 Figure 9 and Figure 7 are some examples on urban, rural and marginal landfills from the GIS model. The GIS data on land use helped identify the location of some of the landfills in purple as seen in Figure 6 and Figure 9, however only a portion of landfills were identified this way. Figure 8 shows a landfill that was not picked up using the land use data but was found manually using google maps and municipal repositories.

The data package for population density was overlayed and contrast increased. This resulted in dark patches on the layouts which indicated larger population density. Landfills were classified as urban if they showed significant infrastructure within 3-5kms of the site as well and dark population patches as seen in Figure 6.

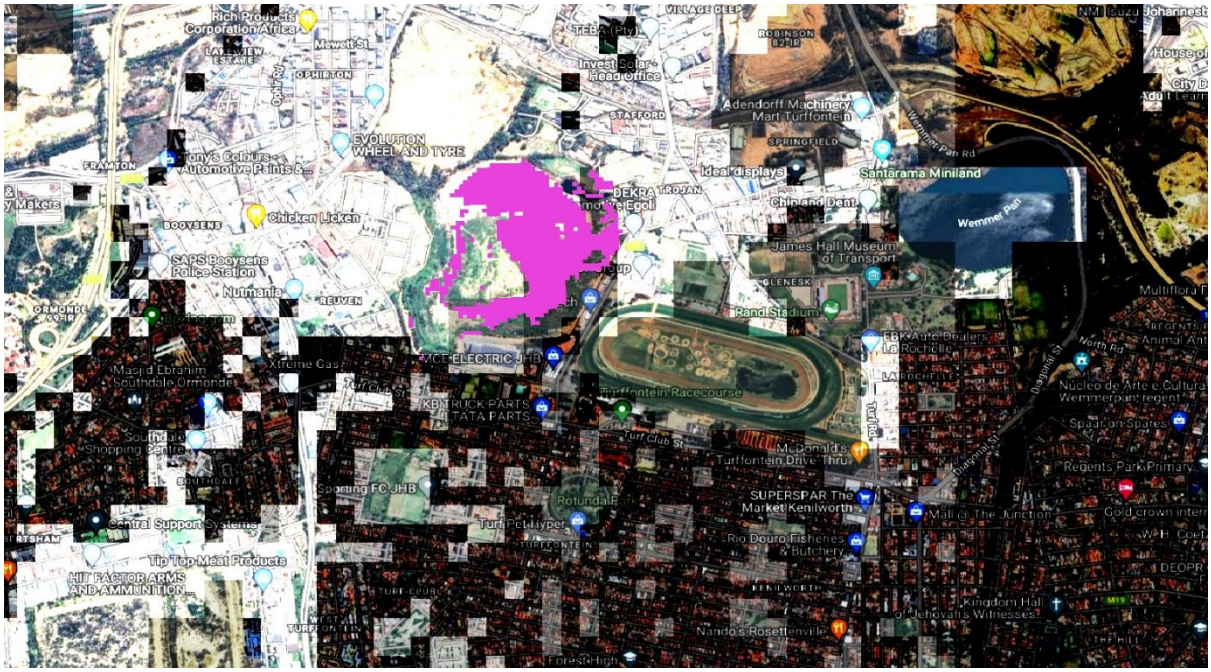

Figure 6: Robinson Deep Landfill site in Gauteng - Urban centre

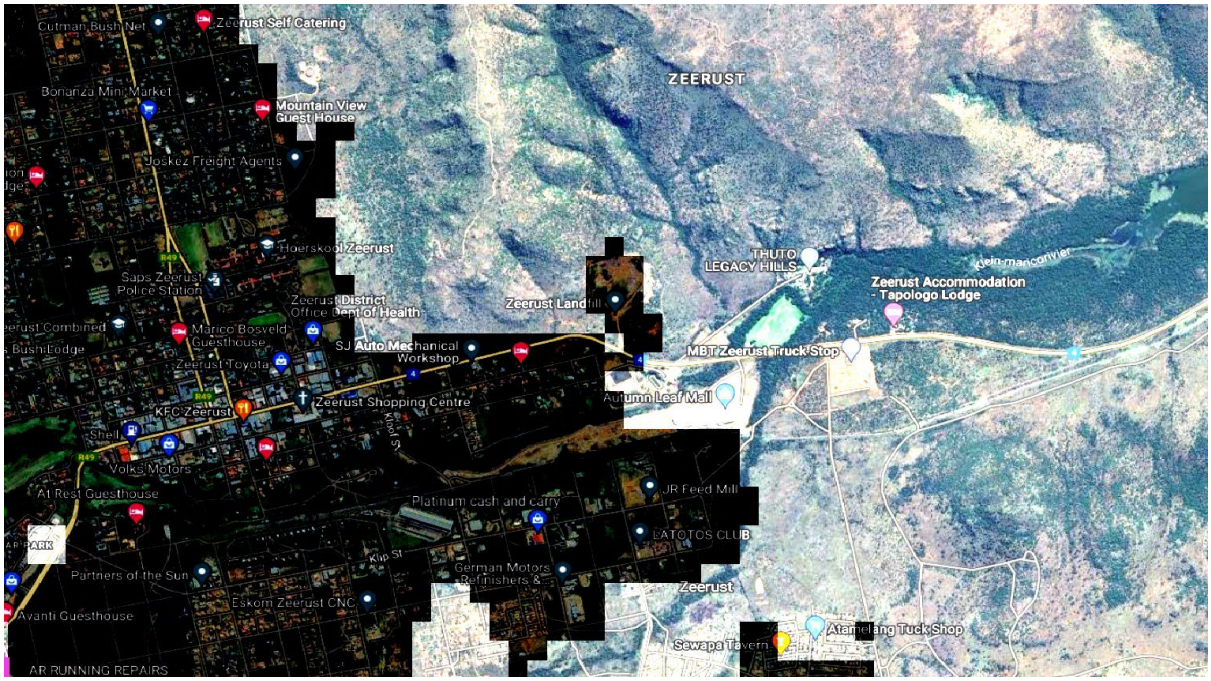

Figure 7: Zeerust Landfill site in North West - Marginal Urban

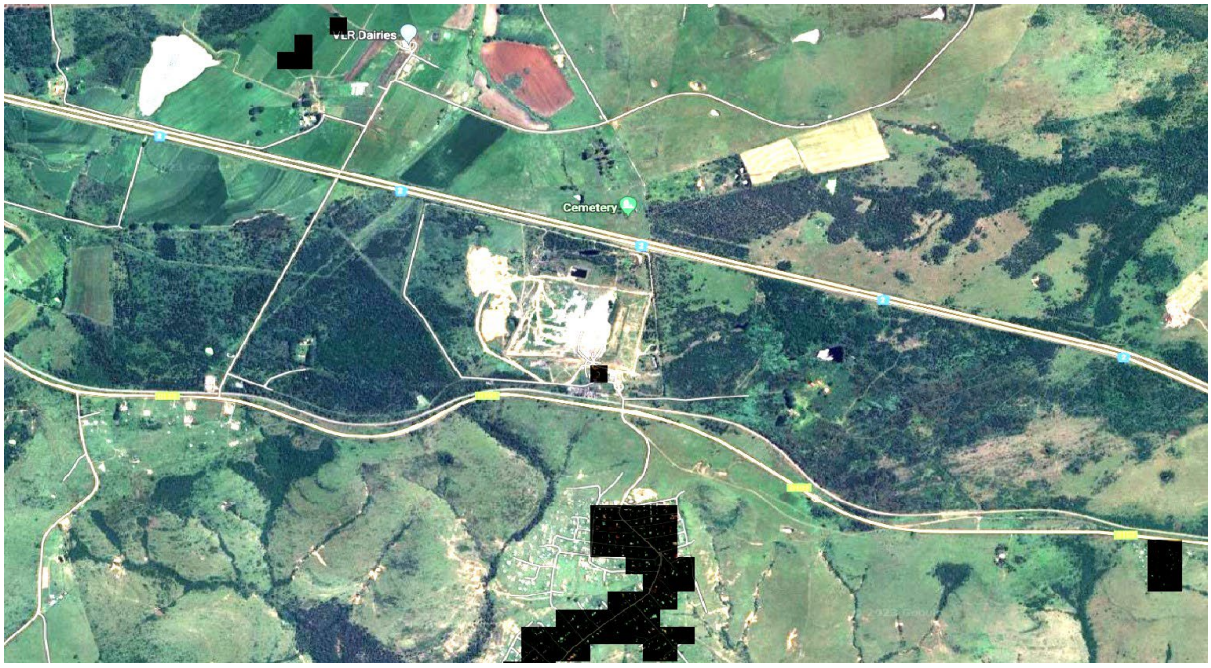

Figure 8: Roundhill Landfill Site in Eastern Cape - Marginal rural area

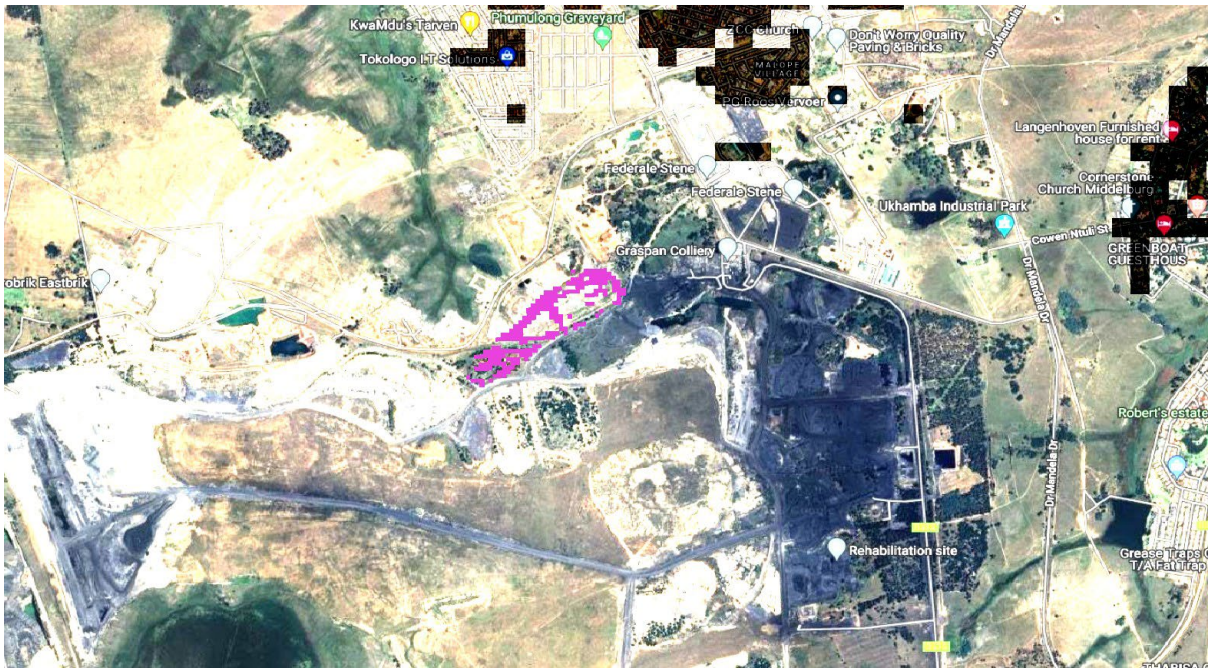

Figure 9: Belfast Landfill site in Mpumalanga - Rural

5.3.7 Disamenity Cost

The disamenity costs summarised in Table 21 above from section 5.2 is dependent on the location to population and compliance. Once the landfills had been categorised as rural or urban the disamenity cost could be attributed. Using the landfill audit reports by AfriForum, the landfill compliance could be assessed (De Klerk, 2023). A compliance rating of under 80% meant it did not comply with minimum requirements as detailed in its waste use license, based on this the landfill

is assumed to be non-compliant and have a higher disamenity cost. Items used to audit the landfills included access control and security, state of infrastructure and plant, site operations, daily cover and nuisance control, drainage and monitoring. All these items contribute to reducing the disamenity of the site. (De Klerk, 2023)

5.3.8 Summation

The total external cost/benefit of landfilling in South Africa can be determined using the total external benefit equation:

$$B_E = \sum_{n=0}^{N}[(PE_n\text{-}NE_n)] \quad (3)$$

A spreadsheet was developed with the listed landfills, with their assumptions as per the above sections and the status quo external cost of landfilling in South Africa is determined. There is no external benefit attributed to landfilling and any benefit due to electricity production and offsetting is factored into the costing as discussed previously. An example of the calculation for Gauteng province can be seen in Table 24, where the total external cost for the status quo is estimated at R125.70 per tonne of MSW. This is derived from the weighted average of each landfill site, weighted by its size, as discussed in 5.3.3.

To get a national external cost, the external cost for each province is summed using the same weighted average technique based on the volume of waste each province produces as seen in Table 16 from section 3.1.2 in this report. The external cost to landfill in South Africa is estimated to be R135.07 per tonne of MSW landfilled. A summary per province can be seen in Table 25: Summary of the results for status quo landfill externalities of 158 landfills in South Africa, weighted by province.

Table 24: Snip of the external benefit calculation for the Gauteng Province

Province: Gauteng

Landfill site name	2023 Compl. (%)	Landfill Size	Water Class	Climate Region	LFG Capture (Yes/No)	LFG capture with ER (Yes/No)	Location	GHG Cost (R/tonne MSW)	LAP Cost (R/tonne MSW)	Leachate Cost (R/tonne MSW)	Disamenity Cost (R/tonne MSW)	**Total external cost**	**Weighted External Cost**
Bon Accord	70	L	B-	Wet Temperate	No	No	Urban	96.47	1.86	0.05	10.52	**108.90**	**6,534.13**
Weltevreden	70	M	B-	Wet Temperate	Yes	No	Urban	40.20	1.50	0.05	105.23	**146.98**	**3,821.57**
Bronkhorstspruit	78	M	B-	Wet Temperate	No	No	Urban	96.47	1.86	0.05	105.23	**203.61**	**5,293.92**
Carletonville	59	M	B-	Wet Temperate	No	No	Rural	96.47	0.57	0.05	52.62	**149.72**	**3,892.66**
Ga-Rankuwa	75	M	B-	Wet Temperate	No	No	Urban	96.47	1.86	0.05	105.23	**203.61**	**5,293.92**
Simmer & Jack	41	L	B-	Wet Temperate	Yes	Yes	Urban	-	2.65	0.05	105.23	**107.94**	**6,476.16**
Hatherley	72	L	B-	Wet Temperate	No	No	Urban	96.47	1.86	0.05	105.23	**203.61**	**12,216.73**
Luipaards Vlei	92	M	B-	Wet Temperate	No	No	Urban	96.47	1.86	0.05	105.23	**203.61**	**5,293.92**
Meyerton	94	M	B-	Wet Temperate	No	No	Rural	96.47	0.57	0.05	5.26	**102.36**	**2,661.30**
Mooiplaats	16	L	B-	Wet Temperate	No	No	Rural	96.47	0.57	0.05	5.26	**102.36**	**6,141.45**
Randfontein	84	S	B-	Wet Temperate	No	No	Urban	96.47	1.86	0.05	105.23	**203.61**	**1,425.29**
Marie Loiuse	74	L	B-	Wet Temperate	Yes	No	Urban	40.20	1.50	0.05	10.52	**52.27**	**3,136.40**
Soshanguve	72	M	B-	Wet Temperate	No	No	Urban	96.47	1.86	0.05	105.23	**203.61**	**5,293.92**
Rietfontein	18	L	B+	Wet Temperate	Yes	No	Urban	40.20	1.50	0.03	105.23	**146.96**	**8,817.36**
Boitshepi)	93	M	B-	Wet Temperate	No	No	Urban	96.47	1.86	0.05	105.23	**203.61**	**5,293.92**
Robinson Deep	94	L	B-	Wet Temperate	Yes	Yes	Urban	-	2.65	0.05	10.52	**13.23**	**793.56**
Ennerdale	93	C	B-	Wet Temperate	Yes	No	Urban	40.20	1.50	0.05	10.52	**52.27**	**52.27**
Goudkoppies	72	L	B+	Wet Temperate	Yes	No	Urban	40.20	1.50	0.03	10.52	**52.25**	**3,134.76**
Platkop	70	L	B-	Wet Temperate	No	No	Rural	96.47	0.57	0.05	52.62	**149.72**	**8,983.05**
Total external impacts								65.75	1.59	0.05	57.68		**125.07**

Table 25: Summary of the results for status quo landfill externalities of 158 landfills in South Africa, weighted by province.

Province	Number of landfill sites reviewed in this study	Waste generated as a percentage of total waste (%)	Cost of landfill externalities (R/tonne waste)
Gauteng	19	14.7	R125.07
KwaZulu-Natal	10	10.5	R162.33
Limpopo	14	1.1	R160.71
Mpumalanga	20	10.5	R153.13
Northern Cape	17	17.9	R134.67
Northwest	21	6.3	R101.62
Eastern Cape	8	26.3	R124.10
Freestate	26	4.2	R194.11
Western Cape	23	8.4	R79.98
Weighted average	158	100	**R135.07**

CHAPTER 6: COST BENEFIT ANALYSIS: TECHNOLOGY COMPARISON

This chapter intends to apply a cost benefit analysis for each potential technology to derive the total cost for maximum benefit from food waste. Each technology will be analysed separately in this section and the total benefit will be determined.

For each technology identified as having potential benefit to processing food waste the below criterion needs to be accounted for each section:

Internal Costs: Pre-treatment and treatment cots (OpEx and Capex)

Internal Benefit: Gate fees, energy production, marketable by-products

External Costs: Greenhouse gas production, local air pollutants, emissions to ground (Leachate), disamenity cost.

External Benefits: Emission displacement, landfill diversion (saving on OpEx, GHG reduction, leachate reduction, local air pollutant reduction, disamenity reduction), reduced fertilizer use.

6.1 Landfilling (status quo)

6.1.1 Object Definition

The object of this scenario is to quantify the Total Benefit (B_T) for the current status quo landfilling of waste in 158 landfills across South Africa. This will be used as a benchmark to compare against other potential technologies for food waste processing options. The objective function to be optimised is shown in the equation below:

$$B_T = B_P + B_E$$

$$B_T = \sum_{n=0}^{N}[(AVW_n \times SP)\text{-}(IC_n + OMC_n) + (PE_n\text{-}NE_n)] \quad (4)$$

$$B_{T-Land} = \left(AVW_{landfill} \times SP_{landfill}\right) - \left(IC_{landfill} + OMC_{landfill}\right) + \left(PE_{landfill} - NE_{landfill}\right) \quad (5)$$

6.1.2 Project Impacts

In this section the key is the accurate identification, frequency, quantification, and monetary evaluation of the impacts (private and external) of status quo landfilling of food waste. In Table 26, a summary of the impacts is specified. The principal characteristics of each impact group analysed are detailed below.

Table 26: Summary of the status quo landfilling of waste impacts.

Type of cost	Impact Group	Description of impact	Impact Identification	
			Cost	Revenue
Internal	Infrastructure	Treatment and final disposal	OpEx and CapEx	
Internal	Recovery of waste	Gate Fees		Market Price
Internal		Energy		Market Price
External	Environmental	Emissions to air	Greenhouse gases	
External		Emissions to air	Local air pollutants	
External		Emissions to ground	Leachate	
External	Quality of life	Disamenities	Disamenity	

6.1.3 Infrastructure

This impact group includes the private costs related to the infrastructure for operation, maintenance, and capital expenditure (Capex and Opex). Table 9 in Chapter 2: Literature Review, section 2.3.1.1, details these costs from the literature. This section estimates the Capex and Opex costs of landfilling to be R280.00 for small landfills and R507.00 per tonne of waste for medium and large sites.

Further the same private cost for LFG collection without energy production and with energy production is estimated to be R564.00 and R597.30 per tonne of waste respectively. By assigning these values to each of the 158 landfills in this study, depending on their state, a Capex and Opex cost of R483.27 per tonne of waste is calculated, as seen in Table 27.

Table 27: Summary of results for status quo landfill private cost of 158 landfills in South Africa, weighted by provincial waste production.

Province	Number of landfill sites reviewed in this study	Waste generated as a percentage of total waste (%)	Landfill Capex and Opex Costs (R/tonne waste)
Gauteng	19	14.7	534.84
KwaZulu-Natal	10	10.5	462.54
Limpopo	14	1.1	454.62
Mpumalanga	20	10.5	449.40
Northern Cape	17	17.9	471.62
Northwest	21	6.3	418.98
Eastern Cape	8	26.3	490.64
Freestate	26	4.2	438.29
Western Cape	23	8.4	507.08
Weighted total	158	100	**484.67**

6.1.4 Recovery from waste

6.1.4.1 Gate Fees

Table 11 in Chapter 2, calculates an average weighted tariff based on all municipal gate fees, weighted by the volume of waste each municipality produces, estimating this to be R376.28 per tonne of waste.

$SP_{landfill}$ = 376.28

6.1.4.2 LFG-to-Energy

Of the 158 landfills under review only two produce electricity from landfill gas and are still operational, these are; Simmer and Jack and Robinson Deep, both located in Gauteng Province. Neither of these LFG-to-energy projects receive direct income from the landfill but 'offset' the municipal bill with Eskom. The exact saving is unclear, but from the REIPPP determination issued by NERSA, the average price of energy production from renewables in FY20/21 was R2.10 kWh. (NERSA, 2022)

Further, based on the case study by Franks et al. (2014) on the Simmer and Jack LFG-to-energy project:

Simmer and jack:

- 1MW, average electrical output 7,135 MWh/year (Franks et al., 2014)
- 430,000 tonnes per year (Mbazima et al., 2022)

Assuming the same production for Robinson deep due to lack of data:

Robinson Deep:

- 4.4 MW, assumed electrical output (4.4 x 7,135 MWh/year) = 31,394 MWh/year
- Average waste disposed over FY15/16 to FY19/20 period: 444,813 tonnes/year (Pikitup, 2021)

From the above, and using a rate of R2.10 kWh, the total estimated income from LFG to energy would be R80,910,900.00 per year, which equates to: R2.23 per tonne of MSW for South Africa.

6.1.5 External impacts

The total external impact has been calculated in chapter 5.3: Pricing landfill externalities across South Africa, estimating a total external cost of R135.07 per tonne of waste. This includes weighted averages for each of the 158 active landfills for their GHG emissions, local air pollutants, leachate production and disamenity cost. No positive impacts are associated with landfilling.

$$B_E = \sum_{n=0}^{N}[(PE_n\text{-}NE_n)] \quad (3)$$

$B_{E\text{-land}}$ = - R135.07 per tonne of waste

6.1.6 Total Benefit

With the impacts being quantified, along with the cost and revenues, it is now possible to calculate the total benefit. Firstly, it needs to be determined if the project is financially viable, this is defined as $B_p > 0$. The private benefit is calculated using:

$$B_P = \sum_{n=0}^{N}[(AVW_n \times SP) - (IC_n + OMC_n)] \quad (6)$$

Table 28 summarises the results of the analysis and the private benefit is calculated as:

$AVW_{landfill}$ = 36,284,479.84 tonnes per year

$B_{p\text{-land}}$ = 13,733.61 - 17,535.35
= - R3,801.74 million per year

Since $B_p < 0$, this suggest the current status quo is not financially viable.

The total benefit, calculated with the total benefit equation below, can be determined as follows:

$B_{T\text{-land}} = B_{P\text{-land}} - B_{E\text{-land}}$
= 13,733.61 – 22,436.29
= - R8,753.18 million per year

6.1.7 Sensitivity analysis: Compliance

A quick analysis was done on the status quo cost if it was assumed that all landfills were compliant. Due to the relatively high externality of disamenities this is thought to have a significant effect on the total external cost by reducing the disamenity cost. Disamenity (compliant status quo) reduces from 50.15 to R7.86 per tonne waste. The total external benefit then reduces to R92.78 per tonne of waste.

$B_{T\text{-land}}$ (compliant status quo) = -R7,218.72 million per year

Saving R1,534.45 million per year.

Table 28: Summary of the economic results of impacts considered for the current status quo waste landfilling in South Africa per ZAR/year and ZAR/tonne (2022 ZAR).

Type of cost	Impact Group	Description of impact	Impact Identification		Impact Quantification (tonnes/year)	Impact Valuation	
			Cost (R/tonne /year)	Revenue (R/tonne /year)		Cost (Million R/year)	Revenue (Million R/year)
Internal	Infrastructure	Treatment and final disposal	484.67	-	36,284,479.84	17,585.84	-
Internal	Recovery of waste	Gate Fees	-	376.28	36,284,479.84	-	13,653.12
Internal		Energy (status quo)	-	2.23	36,284,479.84	-	80.48
External	Environmental	GHG emissions	83.68	-	36,284,479.84	3,036.42	-
External		LAP emissions	1.19	-	36,284,479.84	43.13	-
External		Leachate	0.04	-	36,284,479.84	1.59	-
External	Quality of life	Disamenities	50.15 (7.86)	-	36,284,479.84	1,819.81	-
Total of Internal Impacts						**17,585.84**	13,733.61
Total External Impacts						4,900.94	0.00
Total Impacts						**22,486.78**	**13,733.61**
Assuming 100% Compliance							
Total of Internal Impacts						17,585.84	13,733.61
Total External Impacts						3,366.49	0.00
Total Impacts						**20,952.33**	**13,733.61**

6.2 Landfilling with biogas capture

6.2.1 Object Definition

The object of this scenario is to quantify the Total Benefit (B_T) for the 158 landfills across South Africa if LFG capture (LFGC) systems were to be installed.

The objective function to be optimised is shown in the equation below:

$$B_T = B_P + B_E$$

$$B_T = \sum_{n=0}^{N}[(AVW_n \times SP)\text{-}(IC_n + OMC_n) + (PE_n\text{-}NE_n)] \quad (4)$$

$$B_{T-LFGC} = (AVW_{LFGC} \times SP_{LFGC}) - (IC_{LFGC} + OMC_{LFGC}) + (PE_{LFGC} - NE_{LFGC}) \quad (7)$$

6.2.2 Project Impacts

The project impacts are listed in Table 29, quantification, and monetary evaluation of the impacts (private and external) for LFG collection systems will follow.

Table 29: Summary of the impacts of LFG collection systems.

Type of cost	Impact Group	Description of impact	Impact Identification	
			Cost	Revenue
Internal	Infrastructure	Treatment and final disposal	OpEx and CapEx	
Internal	Recovery of waste	Gate Fees		Market Price
Internal		Energy		Market Price
External	Environmental	Emissions to air	Greenhouse gases	
External		Emissions to air	Local air pollutants	
External		Emissions to ground	Leachate	
External	Quality of life	Disamenities	Disamenity	

6.2.3 Infrastructure

As indicated previously, Table 9 in Chapter 2, section 2.3.1.1, details these costs from the literature and can be summarised as follows:

OpEx and CapEx landfill only: R507.00 and 280.00 /tonne waste
OpEx and CapEx LFG collection: R564.00 /tonne waste

As discussed in section 2.3.1.1, only medium and large landfills are considered viable for LFG collection, small and communal landfills are assumed to remain unchanged. Table 30 summarises the private costs per province based on the above assumptions. The weighted private cost is R522.82 per tonne of waste.

Table 30:Summary of results for LFG collection private cost of 158 landfills in South Africa, weighted by provincial waste production.

Province	Number of landfill sites reviewed in this study	Waste generated as a percentage of total waste (%)	Landfill Capex and Opex Costs (R/tonne waste)
Gauteng	19	14.7	560.99
KwaZulu-Natal	10	10.5	508.37
Limpopo	14	1.1	498.46
Mpumalanga	20	10.5	491.94
Northern Cape	17	17.9	519.74
Northwest	21	6.3	453.88
Eastern Cape	8	26.3	543.53
Freestate	26	4.2	478.04
Western Cape	23	8.4	541.81
Weighted total	158	100	R522.82

6.2.4 Recovery from waste

6.2.4.1 Gate Fees

It is assumed that the gate fee remains the same as LFG management is a minimum requirement is South Africa and is assumed to be factored into municipal lifecycle costing.

SP_{LFGC} = 376.28

6.2.4.2 LFG-to-Energy

The same assumptions are applied as in 6.1.4.2, and only two of the 158 sites with active LFG capture are considered.

6.2.5 External impacts

By applying the same methodology as in section 5.3: Pricing landfill externalities across South Africa, an external cost can be determined for each of the 158 landfills.

With reference to Table 18, which summarises the external costs, each landfill is given an external cost according to whether it has LFG capture.

Table 31 summarises this analysis and gives an externality cost of R87.25 per tonne of waste.

Table 31: Summary of the results for LFG collection externalities of 158 landfills in South Africa, weighted by provincial waste production.

Province	Number of landfill sites reviewed in this study	Waste generated as a percentage of total waste (%)	Cost of landfill externalities (R/tonne waste)
Gauteng	19	14.7	83.22
KwaZulu-Natal	10	10.5	96.92
Limpopo	14	1.1	98.85
Mpumalanga	20	10.5	99.44
Northern Cape	17	17.9	91.51
Northwest	21	6.3	81.00
Eastern Cape	8	26.3	64.99
Freestate	26	4.2	134.23
Western Cape	23	8.4	57.26
Weighted total	158	100	**87.25**

It is important to note that many of the landfills are still considered non-compliant and this will have considerable effect on the disamenity cost. Hence it is proposed that two cases are scrutinised, current compliance and assumed 100% compliance for all sites.

6.2.6 Total Benefit

The total benefit as seen in the summary of calculations in Table 32 is:

$B_{T\text{-}LFGC}$ = - R8,402.32 million per year (status quo compliance)

$B_{T\text{-}LFGC}$ = - R7,181.83 million per year (100% compliance)

Table 32: Summary of the economic results of impacts considered for LFG capture per ZAR/year and ZAR/tonne (2022 ZAR), with varying compliance.

Type of cost	Impact Group	Description of impact	Impact Identification		Impact Quantification (tonnes/year)	Impact Valuation	
			Cost (R/tonne /year)	Revenue (R/tonne /year)		Cost (Million R/year)	Revenue (Million R/year)
Internal	Infrastructure	Treatment and final disposal	522.82		36,284,479.84	18,970.17	-
Internal	Recovery of waste	Gate Fees		376.28	36,284,479.84	-	13,653.12
Internal		Energy (status quo)		2.22	36,284,479.84	-	80.48
External	Environmental	GHG emissions	46.19		36,284,479.84	1,675.83	-
External		LAP emissions	1.07		36,284,479.84	38.90	-
External		Leachate	0.04		36,284,479.84	1.59	-
External	Quality of life	Disamenities	39.95 (6.31)		36,284,479.84	1,449.44	-
Total of Internal Impacts						18,970.17	13,733.61
Total External Impacts						3,165.76	0.00
Total Impacts						**22,135.93**	**13,733.61**
Assuming 100% Compliance							
Total of Internal Impacts						18,970.17	13,733.61
Total External Impacts						1,945.41	0.00
Total Impacts						**20,915.58**	**13,733.61**

6.3 Landfilling with biogas capture and energy recovery

6.3.1 Object Definition

The object of this scenario is to quantify the Total Benefit (B_T) for the 158 landfills in South Africa if LFG capture systems were to be installed and electricity was produced.

$$B_{T\ LFG-E} = (AVW_{LFG-E} \times SP_{LFG-E}) - (IC_{LFG-E} + OMC_{LFG-E}) + (PE_{LFG-E} - NE_{LFG-E}) \quad (8)$$

6.3.2 Project Impacts

The project impacts are listed in Table 33, quantification, and monetary evaluation of the impacts (private and external) for LFG collection systems will follow.

Table 33: Summary of the impacts of LFG capture and electricity production.

Type of cost	Impact Group	Description of impact	Impact Identification	
			Cost	Revenue
Internal	Infrastructure	Treatment and final disposal	OpEx and CapEx	
Internal	Recovery of waste	Gate Fees		Market Price
Internal		Energy		Market Price
External	Environmental	Emissions to air	Greenhouse gases	
External		Emissions to air	Local air pollutants	
External		Emissions to ground	Leachate	
External	Quality of life	Disamenities	Disamenity	

6.3.3 Infrastructure

As indicated previously, Table 9 in Chapter 2, section 2.3.1.1, details these costs from the literature and can be summarised as follows:

OpEx and CapEx landfill only: R507.00 and 280.00 /tonne waste
OpEx and CapEx LFG capture and electricity recovery: R597.30 /tonne waste

As discussed previously, only medium and large landfills are considered viable, small and communal landfills are assumed to remain unchanged. Table 34

summarises the private costs per province based on the above assumptions. The weighted private cost is R551.29 per tonne of waste.

Table 34: Summary of results for LFG collection with electricity recovery private cost of 158 landfills in South Africa, weighted by provincial waste production.

Province	Number of landfill sites reviewed in this study	Waste generated as a percentage of total waste (%)	Landfill Capex and Opex Costs (R/tonne waste)
Gauteng	19	14.7	593.94
KwaZulu-Natal	10	10.5	535.15
Limpopo	14	1.1	524.08
Mpumalanga	20	10.5	516.79
Northern Cape	17	17.9	547.85
Northwest	21	6.3	474.27
Eastern Cape	8	26.3	574.43
Freestate	26	4.2	501.26
Western Cape	23	8.4	572.51
Weighted total	158	100	551.29

6.3.4 Recovery from waste

6.3.4.1 Gate Fees

It is assumed that the gate fee remains the same. Gate fees: R376.28 /tonne waste.

$SP_{LFG\text{-}E}$ = 376.28

6.3.4.2 LFG-to-Energy

Assuming all medium and large landfills are receiving LFG-to-energy and landfill life is 30 years and 50 years after closure. Table 35 lists the parameters used to estimate the potential value derived per tonne of waste sent to landfill. This is estimated to be R258.19 per tonne of wet waste. This falls within estimates proposed in the GreenCape Report (2022b) of value potential of biogas per tonne of input organics in the range of R48-R726.

In addition to this, when calculating the energy recovery potential of landfill gas, by multiplying the calculated 516.47 MJ/tonne by the estimated volume of organic waste landfilled in a year, a value of 18,738.81 TJ/year is determined which is of the same order of magnitude of landfill gas potential of 22,710 TJ/year estimated by Scarlat et al. (2015).

Table 35: Parameters used for LFG energy production estimate from literature.

Parameters		Comment	Reference
RSA methane emissions without gas collection (kg tonne^{-1} wet waste)	44.29	Adjusted from RSA study and waste contents, by adjusting the DOC to match revised waste volumes.	(Friedrich & Trois, 2013a)
LFG collection efficiency (%)	60	The collection efficiency varied widely with literature; Friedrich & Trois (2013a) use 75% in the RSA context. BDA Group estimate will be used to match assumptions.	(BDA Group, 2009)
Energy content of methane (MJ/kg)	55.52	Friedrich & Trois (2013a) use 51.91. BDA Group estimate will be used to match assumptions.	(BDA Group, 2009)
Efficiency of methane conversion to electricity (%)	30	Matching assumptions in literature across studies.	(BDA Group, 2009) (Friedrich & Trois, 2013a)
Conversion factor (MJ/kWh)	3.6		(BDA Group, 2009)
Price per kWh (R/kWh)	2.10	NERSA average price for renewables in FY20/21	(NERSA, 2022)

It is to be noted that when calculating the total private benefits, the energy benefit is only to be multiplied by the volume of waste going to the large and medium landfills. Using the weighted averages method and assuming zero energy benefit is applied to small and communal landfills, this is further revised down to R220.75.

6.3.5 External impacts

By applying the same methodology as in section 5.3: Pricing landfill externalities across South Africa, an external cost can be determined for each of the 158 landfills. With reference to Table 18, which summarises the external costs, each medium and large landfill is given an external cost assuming LFG capture and electricity generation, while smaller landfills are assumed to remain as they are.

Table 36 summarises this analysis and gives an externality cost of R53.23 per tonne of waste.

Table 36: Summary of the results for LFG collection and electricity production externalities of 158 landfills in South Africa, weighted by provincial waste production.

Province	Number of landfill sites reviewed in this study	Waste generated as a percentage of total waste (%)	Cost of landfill externalities (R/tonne waste)
Gauteng	19	14.7	50.61
KwaZulu-Natal	10	10.5	58.64
Limpopo	14	1.1	61.67
Mpumalanga	20	10.5	69.38

Northern Cape	17	17.9	57.49
Northwest	21	6.3	56.14
Eastern Cape	8	26.3	27.31
Freestate	26	4.2	106.84
Western Cape	23	8.4	20.52
Weighted total	158	100	**53.23**

It is important to note that many of the landfills are still considered non-compliant and this will have considerable effect on the disamenity cost. Hence two cases are scrutinised, current compliance and assumed 100% compliance for all sites.

6.3.6 Total Benefit

The total benefit as seen in the summary of calculations in Table 37 is:

$B_{T\ LFG\text{-}E}$ = -R271.68 million per year (status quo compliance)

$B_{T\ LFG\text{-}E}$ = R948.81 million per year (100% compliance)

Table 37: Summary of the economic results of impacts considered for LFG capture and electricity production per ZAR/year and ZAR/tonne (2022 ZAR), with varying compliance.

Type of cost	Impact Group	Description of impact	Impact Identification		Impact Quantification (tonnes/year)	Impact Valuation	
			Cost (R/tonne /year)	Revenue (R/tonne /year)		Cost (Million R/year)	Revenue (Million R/year)
Internal	Infrastructure	Treatment and final disposal	551.29		36,284,479.84	20,003.24	-
Internal	Recovery of waste	Gate Fees		376.28	36,284,479.84	-	13,653.12
Internal		Energy		220.75	36,284,479.84	-	9,346.16
External	Environmental	GHG emissions	11.82		36,284,479.84	428.91	-
External		LAP emissions	1.42		36,284,479.84	51.43	-
External		Leachate	0.04		36,284,479.84	1.59	-
External	Quality of life	Disamenities	39.95 (6.31)		36,284,479.84	1,449.44	-
Total of Internal Impacts						20,003.24	21,662.94
Total External Impacts						1,931.36	0.00
Total Impacts						**22,135.93**	**21,662.94**
Assuming 100% Compliance							
Total of Internal Impacts						20,003.24	21,662.94
Total External Impacts						711.01	0.00
Total Impacts						**20,714.25**	**21,662.94**

6.4 Windrow Composting

6.4.1 Object Definition

The object of this scenario is to quantify the Total Benefit (B_T) of diverting food waste from 158 landfills in South Africa to produce compost using open windrow composting.

$$B_{T\,Comp} = \left[\left(AVW_{comp} \times SP_{comp}\right) - \left(IC_{comp} + OMC_{comp}\right) + \left(PE_{comp} - NE_{comp}\right)\right] + \left[\left(AVW_{land\,C} \times SP_{land\,C}\right) - \left(IC_{land\,C} + OMC_{land\,C}\right) + \left(PE_{land\,C} - NE_{land\,C}\right)\right] \quad (9)$$

6.4.2 Project Impacts

The project impacts are listed in Table 38, quantification, and monetary evaluation of the impacts (private and external) for LFG collection systems will follow.

Table 38: Summary of the impacts of food waste diversion to windrow composting.

Type of cost	Impact Group	Description of impact	Impact Identification	
			Cost	Revenue
Internal	Infrastructure	Treatment and final disposal	OpEx and CapEx	
Internal	Recovery of waste	Gate Fees		Market Price
Internal		Compost		Market Price
External	Use of materials	Landfill diversion		Avoid material sent to landfill
	Environmental	Replace chemical fertiliser		Reduces nitrogen fert. use + CO_2 sequestration
External		Emissions to air	GHG production	Reduction in landfill GHG
External		Emissions to air	LAP production	Reduction in landfill LAP
External		Emissions to ground	Leachate	Reduction in landfill leachate
External	Quality of life	Disamenities	Disamenity	

6.4.3 Infrastructure

As indicated previously, Table 13: Expected cost ranges for various AWT options in Chapter 2, details these costs from the literature and suggests an OpEx and CapEx cost for composting of R308 per tonne of organic waste processed.

Landfill private costs are assumed to stay the same as the status quo scenario, however the waste volumes processed will reduce as the food waste will be assumed to be diverted for alternative use.

6.4.4 Recovery from waste

6.4.4.1 Gate Fees

It is assumed that the gate fee remains the same as landfilling and composting, however this is an underestimate. Most landfills receiving large volumes of putrescible waste would require a safe disposal which costs considerably more. Table 11 indicates a weighted average fee of R1,014.47 per tonne of putrescible waste.

$SP_{land\ C} = SP_{comp}$ = R376.28 per tonne waste.

6.4.4.2 By-Product: Compost

By diverting food waste from landfill for composting there is an additional benefit of revenue from compost sales, however the quality of compost can vary considerably and hence the revenue. Quality of windrow composting can be significantly lower than other composting processes (Liu et al., 2020).

Given the average compost density of 548kg/m3 (The US Composting Council, 2021) and South Africa's average bulk compost prices of around R530/m3, the price equates to R967.15 per tonne of compost. However, only 250kg of compost can be derived from a tonne of food waste (Kim et al., 2011), leading to a pre-cost value of R241.78 per tonne for food waste as a composting input. This is in line with figures suggested by GreenCape (2022b) which suggested a range of R75.00-R1980.00 per tonne of organics.

6.4.5 External impacts

6.4.5.1 Reduced waste volume landfilled.

As discussed in the literature review, Table 4 summarises the estimated proportion of landfill waste that is food waste, this is estimated to be 8% of the total waste volume and therefor a reduction of 8% in waste that needs to be landfilled.

6.4.5.2 Greenhouse Gas – Cost

Aerobic composting as a process produces net GHG but also has the potential to have a GHG saving (Friedrich & Trois, 2013b). GHG emission result from the energy required for the facility to operate, machinery (chipper, TLB, etc.) and by the degradation process itself which produces small volumes of methane and nitrous oxides. Benefits from compost will be reviewed in the next section.

The same mass balance assumptions and process used by Friedrich & Trois (2013b) are applied, however replacing the garden waste assumptions with food waste. It is assumed that food waste reduced the production of GHG in compost relative to green waste, and a study by Williams et al. (2019) suggested that by mixing food waste with green waste there was a reduction of 1.4 times the global warming potential (GWP) (Williams et al., 2019), the studies looking at food waste in compost can be seen in Table 39 and from the average 122.79 kg CO_2 e/tonne waste will be used in place of the 236.03 kg CO_2 e/tonne waste estimate by Friedrich & Trois (2013b) for the composting process only. The other GHG emissions due to machinery and operations remains the same.

This gives a total GHG emission external impact of 130.89 kg CO_2 eq per tonne food waste when including machinery emissions.

To have comparative costing between the technologies the same assumptions used in the BDA Group (2009) study need to be applied to get an equivalent social cost of carbon. The BDD Group (2009) study used 40 A$ (in 2008 A$) per tonne CO_2, as well as assuming 7% discount rate over the lifespan of the compost plant.

Applying the equivalent methodology to composting, assuming 30 years of compost production and GHG emissions, with a constant rate of annual emissions over the 30-year period, a 7% discount rate, assuming R564.25 per tonne CO_2 (2008 ZAR – 40 A$ equivalent) gives R31.94 per tonne of waste (2008 ZAR), which is equivalent to R65.00 per tonne of waste (2002 ZAR).

Table 39: GWP of food waste compost compared to green waste compost from literature.

Waste source	GWP (kg CO_2 equ/tonne waste)	Comment	Reference
Kitchen waste	111		(Mertenat et al., 2019)
Mixed food waste	110		(Williams et al., 2019)
Mixed food waste	80	Process emissions only	(Komakech et al., 2015)
Food waste	80	Estimated using Friedrich & Trois (2013b) assumptions	(Lee et al., 2007)
Food waste	97.25	Meta-analysis – only composting, no replacement	(Bernstad & la Cour Jansen, 2012)
Food waste	282	Meta-analysis	(Schott et al., 2016)
Food waste	99.29	No substitution	(Mondello et al., 2017)
Average	**122.79**		

6.4.5.3 Greenhouse Gas – Benefit Compost

There are several benefits of composting that has been reviewed in the literature. A global meta-analysis determined that replacing synthetic nitrogen fertilisers with organic sources of nitrogen was shown to decrease the average effect of N_2O emissions and have a net decrease in GHG emissions (Fan et al., 2023a).

This report looks at the GHG reduction by replacing inorganic fertilizers to soil and the carbon that is sequestered by the soil after the compost is applied (Friedrich & Trois, 2013b). Freidrich & Trois (2013b) when analysing green garden waste compost suggest a savings of 64.65 kg CO_2 eq/tonne waste for windrow composting and 59.54 CO_2 eq/tonne waste for aerated dome. Two meta-analysis, looking at multiple studies on food waste, suggest a reduction of 160 kg CO_2 eq/tonne food waste (Schott et al., 2016) and 100 kg CO_2 eq/tonne food waste (Bernstad & la Cour Jansen, 2012).

The range from the literature is due to sensitivities to assumptions and local conditions. To get an estimate applicable to the South African context, the methodology described by Boldrin et al. (2009), as well as assumptions made by Freidrich & Trois (2013b) for the external benefit of composting will be used but applied to the nutrient output ratios from Komakech et al. (2015) and Boldrin et al. (2009) which looks at food waste rather than green waste.

This calculation is based on Boldrin et al. (2009) paper on compost utilization breaks this into two parts; savings due to the substitution of inorganic fertiliser due to compost application and savings due to carbon bound to soil due to compost application. Carbon bound ($CO_{2,bind}$) in the soil can be determined by:

$$CO_{2,bind} = C_{input} \times C_{bind} \times \frac{44}{12} \qquad (10)$$

(Boldrin et al., 2009)

Where the C fraction bound (C_{bind}) to soil over 100 years has been estimated to be 2-14% (Boldrin et al., 2009). This study will use the same assumption of 8% as Freidrich & Trois (2013b). Carbon input (C_{input}) varies considerable depending on the waste stream and technology, the below assumptions will be used for this study from Komakech et al. (2015) - Table 40.

To determine the displacement of chemical fertilisers using compost assumes the compost is used to replace the N:P:K from chemical sources. An important factor that determines the efficiency of the fertiliser is its utilisation efficiency (I.e. the rate that fertiliser is made available to plants), which is higher for chemical fertiliser and lower for the slow release in organic alternatives (Boldrin et al., 2009). For P and K the Utilization efficiency is assumed to be 100% (Friedrich & Trois, 2013b), while for N this can vary by 20-70% depending on the amendment, soil and crop (Boldrin et al., 2009). The amount of mineral fertiliser displaced ($Mass_{disp}$) depends on nutrient in organic fertiliser (Kg) and substitution efficiency, this is given by:

$$Mass_{disp} = Input_{nutrient} \times Subs_{eff} \qquad (11)$$

(Boldrin et al., 2009)

When comparing the rates of nitrogen release from compost, anaerobic digestate, BSF frass, and vermicastings, especially when considering replacing synthetic nitrogen fertilizers, the substitution efficiency can vary widely based on several factors including the composition of the input material (in this case, food waste), the processing method, and the maturity of the end product. Here are the assumptions made in this study:

Windrow composting: $Subs_{eff,WC}$ = 40% - assuming slow nitrogen release due to low quality compost.
In-vessel-composting: $Subs_{eff,IVC}$ = 50% - assuming higher quality compost.
Anaerobic Digestate: $Subs_{eff,AD}$ = 70% - Faster release than compost due to higher ammonium content.
Vermicompost: $Subs_{eff,VC}$ = 60% - Moderate release due to high microbial activity and small particle size.
BSF Frass: $Subs_{eff,BSF}$ = 60% - Moderate to fast. The nutrient release rate from BSF frass can be quicker than traditional compost due to its finer particle size and the presence of chitin.
From Freidrich & Trois (2013b): 13 kg CO_2 e is emitted per kg N, 3.095 kg CO_2 e is emitted per kg P and 1.53 kg CO_2 e is emitted per kg K (Friedrich & Trois, 2013b).

Table 40: Assumptions and calculations for GHG benefit ratio between varying technologies. (kg per tonne of wet waste)

Compound	Assumed nutrients in waste	Nutrients out: Windrow compost	Nutrients out: in-vessel compost	Nutrients out: AD	Nutrients out: V-compost	Nutrients out: BSF compost
Carbon	180	75.6	81	102.16	74.29	68.24
Nitrogen	6	3.68	3.68	5.95	3.02	2.57
Phosphorous	1.8	1.8	1.8	1.8	1.8	1.8
Potassium	3.4	3.4	3.4	3.4	3.4	3.4
$CO_{2,bind}$		22.18	23.76	30	22	20
$Mass_{disp}$ N		1.47	1.84	4.17	1.81	1.54
Total emission displaced		29.91	34.69	64.98	34.30	30.79
Spreading on land		1.08	1.08	1.08	1.08	1.08
Total benefit CO_2 eq		51.01	57.37	93.90	55.22	49.71

Based on Table 40, and assuming 51.01 kg CO_2 eq/tonne of food waste, discounting over the 30-year period by 7% gives a benefit of R25.33 per tonne of waste (2022 ZAR).

6.4.5.4 Local Air pollutants

There is a great deal in the literature regarding the GHG emissions of composting, as well as the nitrogen losses of this process, with limited information on other emissions. By equating the literature on emissions from composting to BDA Group (2009) emission damage from each pollutant (discounting over 30 year period at 7%) an estimate damage cost for emission from composting is made, but with a low level of confidence.

Any effects on emissions due to machinery, diesel or electricity equates with those emissions from landfilling (Friedrich & Trois, 2013b, 2013a). A meta-analysis by Morrisa et al. (2013) compares aerobic digestion to to other organic waste management technologies, summarised in Table 41, suggests that windrow composting has the same human respiratory efects as LFGTE. Based on this this study will use the same estimate of R1.42 per tonne of food waste.

Table 41: Summarising findings from Morrisa et al. (2013) ranking each technology against each other from a meta-analysis.

Technology	Human Respiratory	Human toxicity	Ground level smog
Anaerobic digestion	1.0	2.2	1.0
Aerobic digestion	2.0	1.3	2.0
LFGTE	2.0	2.3	1.5
LF Flare	3.5	2.5	NR

6.4.5.5 Leachate

Composting facilities, due to the concentrated nutrient load, do have the potential of environmental contamination, any unmanaged leachate that escaped the facility can cause contamination of groundwater if plants are not constructed to minimum standard or mismanaged (Liu et al., 2020). However, since water is consumed during the composting process, it is assumed that any excess water is recirculated and is reused, so no benefit is considered. (Eunomia, 2002)

6.4.5.6 Landfill externality reduction

Since the organic fraction of the landfill directly contributes to the GHG, LAP, leachate and disamenity costs, a reduction in the proportion of organic waste will result in the equivalent reduction of these factors (See chapter 5.1 Benefit Transfer Method, 5.1.2 for assumption). Table 42 summarises the results.

Table 42: Summary of results for reduced externalities due to 8% reduction in organic fraction of 158 landfills in South Africa, weighted by provincial waste production.

Province	Waste generated as a percentage of total waste (%)	Total cost of landfill externalities –Current compliance (R/tonne waste)	Total cost of landfill externalities –100% compliance (R/tonne waste)
Gauteng	14.7	86.32	55.83
KwaZulu-Natal	10.5	112.03	81.21
Limpopo	1.1	110.91	79.02
Mpumalanga	10.5	105.68	73.62
Northern Cape	17.9	92.94	56.64
Northwest	6.3	70.13	56.72
Eastern Cape	26.3	85.65	72.26
Freestate	4.2	133.97	77.34
Western Cape	8.4	55.20	51.35
Weighted total	100	93.22	66.47

6.4.5.7 Disamenity of composting

Windrow composting facilities attract all the disamenities that are typically associated with landfilling (smell, flies, vermin, etc.), and with such a high organic waste component this risk remains high (Liu et al., 2020). To date there is no study that can quantify the disamenity in monetary terms for any kind of composting facility. The effects cannot be ignored, especially since the disamenity cost has a substantial effect on the external costs. (Eunomia, 2002)

Due to the lack of data on this, there is no basis for quantification (Eunomia, 2002), this report assumes that the disamenity of landfills and open windrow composting are the same for compliant landfills. The assumption is made that separation at source happens at the landfills for each of the 158 landfills, and processing takes place there. The disamenity can then be assumed to remain the same and will vary for rural and urban areas along with levels of compliance as seen in Table 43.

Table 43: Variation in disamenity cost and LAP for 31% reduction in OFMSW for compliant and non-compliant landfills, weighted per province.

Province	Waste generated as a percentage of total waste (%)	LAP (ZAR 2022/tonne of waste)	Disamenity Cost – noncompliant (ZAR 2022/tonne of waste)	Disamenity Cost – compliant (ZAR 2022/tonne of waste)
Gauteng	14.7	1.10	39.81	6.43
KwaZulu-Natal	10.5	0.86	39.01	5.65
Limpopo	1.1	0.55	38.18	4.34
Mpumalanga	10.5	0.59	38.48	4.43
Northern Cape	17.9	0.54	42.45	4.24
Northwest	6.3	0.56	19.64	4.30
Eastern Cape	26.3	0.40	18.65	3.63
Freestate	4.2	1.13	66.23	6.62
Western Cape	8.4	0.71	11.48	5.27
Weighted total	100	0.82	34.61	5.43

6.4.6 Total Benefit

The total benefit as seen in the summary of calculations in Table 44 is:

B_{WC}= - R5,999.42 million per year (status quo compliance)

B_{WC} = - R4,940.42 million per year (100% compliance)

Table 44: Summary of the economic assessment of impacts considered for open windrow composting per ZAR/year and ZAR/tonne (2022 ZAR), varying compliance.

Type of cost	Impact Group	Description of impact	Impact Identification		Impact Quantification (tonnes/year)	Impact Valuation	
			Cost (R/tonne /year)	Revenue (R/tonne /year)		Cost (Million R/year)	Revenue (Million R/year)
Internal	Infrastructure	W-Compost O&M	308.00	-	3,030,271.60	933.32	-
Internal		Landfill O&M	484.67	-	33,254,208.23	16,117.17	-
Internal	Recovery of waste	Gate Fees – landfill	-	376.28	33,254,208.23	-	12,512.89
Internal		Gate fees – W-compost	-	376.28	3,030,271.60	-	1,140.23
Internal		Food waste to compost	-	241.78	3,030,271.60	-	732.66
External	Environmental	Chemical fertiliser replacement	-	23.55	3,030,271.60	-	71.36
External		GHG – landfill	57.75	-	33,254,208.23	1,920.57	-
External		GHG – compost	65.00	-	3,030,271.60	196.97	-
External		LAP – landfill	0.82	-	33,254,208.23	27.28	-
External		LAP – Compost	1.42	-	3,030,271.60	4.30	-
External		Leachate – landfill	0.03	-	33,254,208.23	1.00	-
External		Leachate - compost	-	-	3,030,271.60	-	-
External	Quality of life	Disamenity - compost	34.61 (5.43)	-	3,030,271.60	104.89	-
External		Disamenity - landfill	34.61 (5.43)	-	33,254,208.23	1,151.05	-
Total of Internal Impacts						17,050.07	14,385.78
Total External Impacts						3,406.07	71.36
Total Impacts						**20,456.56**	**14,457.15**
Assuming 100% Compliance							
Total of Internal Impacts						17,050.07	14,385.78
Total External Impacts						2,347.07	71.36
Total Impacts						**19,97.56**	**14,457.15**

6.5 In-Vessel Composting

6.5.1 Object Definition

The object of this scenario is to quantify the Total Benefit (B_T) of diverting food waste from 158 landfills in South Africa to produce compost using in-vessel composting (IVC).

6.5.2 Project Impacts

The project impacts are listed in Table 45, quantification, and monetary evaluation of the impacts (private and external) for LFG collection systems will follow.

Table 45: Summary of the impacts of food waste diversion to in-vessel composting.

Type of cost	Impact Group	Description of impact	Impact Identification	
			Cost	Revenue
Internal	Infrastructure	Treatment and final disposal	OpEx and CapEx	
Internal	Recovery of waste	Gate Fees		Market Price
Internal		Compost		Market Price
External	Use of materials	Landfill diversion		Avoid material sent to landfill
	Environmental	Replace chemical fertiliser		Reduces nitrogen fert. use + CO_2 sequestration
External		Emissions to air	GHG production	Reduction in landfill GHG
External		Emissions to air	LAP production	Reduction in landfill LAP
External		Emissions to ground	Leachate	Reduction in landfill leachate
External	Quality of life	Disamenities	Disamenity	

6.5.3 Infrastructure

As indicated previously Table 13 in Chapter 2, details these costs from the literature and suggests an OpEx and CapEx cost for in-vessel composting of R1,106.50 per tonne of organic waste processed.

Landfill private costs are assumed to stay the same as the status quo scenario, however the waste volumes processed will reduce as the food waste will be assumed to be diverted for alternative use.

6.5.4 Recovery from waste

6.5.4.1 Gate Fees

Gate fees landfilling: R376.28 per tonne waste. As per assumption for windrow composting.

6.5.4.2 By-Product: Compost

High quality compost can sell (at bulk prices) for R750-R850/m3. In-vessel compost is assumed to produce higher quality compost of R750 per cube. Giving R1,368.61 per tonne compost. Divide by 4 = R342.15 per ton food waste (250kg compost is made per tonne input). This is in line with figures suggested by GreenCape (2022b) which suggested a range of R75.00-R1980.00 per tonne of organics.

6.5.5 External impacts

6.5.5.1 Reduced waste volume landfilled

As discussed previously, 8% reduction in processed waste.

6.5.5.2 Greenhouse gas – cost

Table 46: GWP of food waste in-vessel compost from literature.

Waste source	GWP (kg CO_2 equ/tonne waste)	Comment	Reference
Food waste	275		(Oldfield et al., 2018)
OFMSW	150	Different waste source	(Colón et al., 2012)
OFMSW	112	Removing transport – not including fert replacement	(Martínez-Blanco et al., 2010)
Food waste	360	No reductions, 91-95% of total energy from transport	(Cabaraban et al., 2008)
Food waste	480	Mostly due to materials (steel)	(Mu et al., 2017)
Average (no outliers)	**179**		

In-vessel composting has many benefits with respect to GHG emissions: it has a 34% lower nitrogen loss when compared to windrow composting (Liu et al., 2020), and is more efficient at removing pathogens in the treated waste. GHG emission result from the energy required for the facility to operate, machinery (chipper) and by the degradation process itself which produces small volumes of methane and nitrous oxides. However, the vessel itself is made from steel and has an inherent GHG associated with this.

Using the average from the literature of 179 kg CO_2 eq per tonne food waste and discounting it over 30 years as done previously, gives R88.90 per tonne of waste (2022 ZAR).

6.5.5.3 Greenhouse gas – benefit

There are several benefits of composting that have been reviewed in the literature. This report looks at the GHG reduction by replacing inorganic fertilizers to soil and the carbon that is sequestered by the soil after the compost is applied (Friedrich & Trois, 2013b). As per Table 40 above, the GHG benefit of 57.37 kg CO_2 eq/tonne of food waste will be used. When discounted this gives R28.49 per tonne of food waste.

6.5.5.4 Local air pollutants

Based on Mu et al. (2017) smog and respiratory effects, all decrease significantly when compared to landfilling, 153.5% and 547.4% respectively. Further the use of filters assists in controlling the VOC, particulate matter and odours. (Lu et al., 2020) The data on this is limited and it is difficult to make a direct comparison as there is the use of electricity and diesel during the operations that need to be considered. For the purposes of comparison, this study will use the assume the LAP is negligible compared to landfilling.

For the remainder of the waste that is to be landfilled, there is a 31% reduction of organic matter due to the food waste diversion. It is assumed that LAP is directly proportional to the proportion of organic matter in the waste body (Nahman, 2011).

6.5.5.5 Leachate

Due to the contained nature of the process, liquid fraction is carefully monitored and recycled into the composting body, therefore no leachate is produced in this system. (Lu et al., 2020)

6.5.5.6 Disamenity

The disamenity is extremely difficult to quantify without any literature or studies to quantify any potential disamenities due to IVC facilities. Due to the diversion of 31% food waste from the landfill, it is again assumed that the disamenity for landfilling is reduced by the same proportion (see Table 43).

The contained nature of the IVC process as well as the fact that is it an aerobic process significantly reduces the odour potential, and limits nuisances like vermin. While the plant itself and traffic due to food waste deliveries to the plant may still provide nuisance to neighbouring communities, this study will assume it will have the same impact as compliant landfills with the 31% organic waste fraction reduction, R5.43 per tonne of food waste (see Table 43). This is the lowest disamenity impact in this study.

6.5.6 Total Benefit

The total benefit as seen in the summary of calculations in Table 47 is:

B_{IVC} = - R8,079.65 million per year (status quo compliance)

B_{IVC} = - R7,109.09 million per year (100% compliance)

Table 47: Summary of the economic assessment of impacts considered for in-vessel composting per ZAR/year and ZAR/tonne (2022 ZAR), varying compliance.

Type of cost	Impact Group	Description of impact	Impact Identification		Impact Quantification (tonnes/year)	Impact Valuation	
			Cost (R/tonne /year)	Revenue (R/tonne /year)		Cost (Million R/year)	Revenue (Million R/year)
Internal	Infrastructure	IVC O&M	1,106.50	-	3,030,271.60	3,353.00	-
Internal		Landfill O&M	484.67	-	33,254,208.23	16,117.17	-
Internal	Recovery of waste	Gate Fees – landfill	-	376.28	33,254,208.23	-	12,512.89
Internal		Gate fees – IVC	-	376.28	3,030,271.60	-	1,140.23
Internal		Food waste to compost	-	342.15	3,030,271.60	-	1,036.81
External	Environmental	Chemical fertiliser replacement	-	28.49	3,030,271.60	-	150.48
External		GHG – landfill	57.75	-	33,254,208.23	1,920.57	-
External		GHG – IVC	88.90	-	3,030,271.60	251.09	-
External		LAP – landfill	0.82	-	33,254,208.23	27.28	-
External		LAP – IVC	-	-	3,030,271.60	2.49	-
External		Leachate – landfill	0.03	-	33,254,208.23	1.00	-
External		Leachate - IVC	-	-	3,030,271.60	-	-
External	Quality of life	Disamenity - IVC	5.43	-	3,030,271.60	104.89	-
External		Disamenity - landfill	34.61 (5.43)	-	33,254,208.23	1,151.05	-
Total of Internal Impacts						19,470.17	14,689.93
Total External Impacts						3,385.75	86.33
Total Impacts						**22,855.91**	**14,776.26**
Assuming 100% Compliance							
Total of Internal Impacts						19,470.17	14,689.93
Total External Impacts						2,415.19	86.33
Total Impacts						**21,885.35**	**14,776.26**

6.6 Anaerobic Digestion with biogas capture

6.6.1 Object Definition

The object of this scenario is to quantify the Total Benefit (B_T) of diverting food waste from 158 landfills in South Africa to an anaerobic digestor (AD), producing electricity from biogas as well as fertiliser.

6.6.2 Project Impacts

The project impacts are listed in Table 48, quantification, and monetary evaluation of the impacts (private and external) for LFG collection systems will follow.

Table 48: Summary of the impacts of food waste diversion and Anaerobic Digestion with electricity production.

Type of cost	Impact Group	Description of impact	Impact Identification	
			Cost	Revenue
Internal	Infrastructure	Treatment and final disposal	OpEx and CapEx	
Internal	Recovery of waste	Gate Fees		Market Price
Internal		Electricity Fertiliser		Market Price
External	Use of materials	Landfill diversion		Avoid material sent to landfill
	Environmental	Replace chemical fertiliser		Reduces nitrogen fert. use + CO_2 sequestration
External		Emissions to air	GHG production	Reduction in landfill GHG Replace Coal electricity
External		Emissions to air	LAP production	Reduction in landfill LAP Replace Coal electricity
External		Emissions to ground	Leachate	Reduction in landfill leachate
External	Quality of life	Disamenities	Disamenity	

6.6.3 Infrastructure

As indicated previously, Table 13: Expected cost ranges for various AWT options in Chapter 2, details these costs from the literature and suggests an OpEx and CapEx cost for composting of R1,153.75 per tonne of organic waste processed.

Landfill private costs are assumed to stay the same as the status quo scenario, however the waste volumes processed will reduce as the food waste will be assumed to be diverted for alternative use.

6.6.4 Recovery from waste

6.6.4.1 Gate Fees

Gate fees landfilling: R376.28 per tonne waste. As per assumption for windrow composting.

6.6.4.2 By-Product: electricity and fertiliser

De Lange & Nahman (2015) summarised the estimated yield of biogas from food waste, based on literature, of 0.296 m^3 biogas/kg food waste. Pérez-Camacho et al., (2018) follow a similar approach, by combining the literature a revised average is 162.35 m^3/tonne food waste. Biogas has combination of methane and carbon dioxide in the ranges of 50-75% and 50%-25% and this vaired considerable with feedstock. Their study assumes 62% methane and 32% CO_2 by volume (Woon et al., 2016), making methane 100.65 m^3/tonne food waste.

Using the same paramaters as in Table 35 above, but with an assumed energy content of methane to be 39.59 MJ/m^3 (using BDA Group (2009) energy content of methane), gives 332.06 kWh/tonne food waste at R697.32 per tonne of food waste.

Komakech et al. (2015) use 50.8 kg biogas per tonne of waste which results in R493.57 per tonne of waste. This falls within estimates proposed in the GreenCape Report (2022b) of value potential of biogas per tonne of input organics in the range of R48-R726.

Determining the volume of food waste digestate (FWD) is not possible without an understanding of the waste stream, approximately 0.57-1.34 tonne of FWD is estimated to be generated for every tonne of food waste (Shao et al., 2023). For a conservative estimate this study will assume 0.57 tonne FWD for every tonne food

waste. There is no available costing for FWD, so this report will assume a low fertlizer cost of R967.15 per tonne, giving R551.28 per tonne of food waste.

6.6.5 External impacts

6.6.5.1 Reduced waste volume landfilled.

As discussed previously, 8% reduction in processed waste.

6.6.5.2 Greenhouse Gas

An overview of the literature is summarised in Table 49. The GHG emissions are estimated to be 71.53 kg CO_2/tonne of waste, based on the literature. This gives a cost of R35.52 per tonne of waste when discounted as discussed previously. For the displacement of fossil fuel power by biogas energy, the literature suggests an average GWP of 338.25 kg CO_2/tonne of waste, which is R167.86 benefit per tonne of food waste. Friedrich & Trois (2013a) propose a methodology to estimate the avoided emissions, using the same principle and 332.06 kWh/tonne food waste, 342 kg CO_2/tonne of waste is calculated. This is in line with the literature and a benefit of R167.86 will be used.

The literature for the benefit of replacing fertilisers is more varied, and this in large part is due to the variance in feedstock, variations in digestate, varied assumptions on carbon sequestration and N_2O release assumptions (Schott et al., 2016). The average from the literature is 95.12 kg CO_2/tonne of waste. Using the calculated value of 93.90 from Table 40 above, which is in line with the average from literature, this is discounted to R46.63 per tonne of food waste.

Table 49: GWP of food waste from literature for anaerobic digestion (kg CO2 equ/tonne waste).

Waste source	Process GWP	GWP fert. sub.	GWP of elec. sub.	Total	Comment	Reference
Mixed waste	70	-5	-80	-20.9	LCA: Not including transport and construction	(Komakech et al., 2015)
Food waste	36	-15	-	-90	LCA: Meta analysis	(Bernstad & la Cour Jansen, 2012)
Food waste	-	-	-	23.98	Replacing elec from (50% coal fossil fuels)	(Woon et al., 2016)

Food waste	19 to 149	-118 to 2.4	-368 to -227	-375 to -75	Replacing high CO2 elec, no transport or construction	(Moller et al., 2009)
Food waste	58 to 71	-340	-318 to -440	-440	LCA Meta analysis: (8-10 studies) mainly fossil fuel replacement	(Schott et al., 2016)
Food waste	-	-	-	-478.10	Net savings from landfill	(Pérez-Camacho et al., 2018)
Organic waste				-250	LCA – meta 5 studies, including displacement of elec and fert	(Morrisa et al., 2013)
OFMSW	45.2			-		(Colón et al., 2012)
Average (no outliers)	**71.53**	**-95.12**	**-338.25**			

6.6.5.3 Local Air pollutants

As with composting, the LAP for landfilling has been adjusted to the lower organic fraction. For the impact of AD it is difficult to quantify, but a meta-analysis of 82 studies done by Morrisa et al. (2013) compares aerobic composting, landfill gas to energy, anaerobic digestion and landfill gas flaring. From the analysis (see Table 41) of the literature AD ranks higher on the human respiratory index which suggests better LAP than landfilling and aerobic composting. It would be assumed that the higher efficiency of gas collection and treatment would be the reason for this.

It is therefore assumed that the impact of AD on LAP is the same as landfill gas collection with electricity production reduced by 61.2% (98%/60%) for adjusting from 60% collection efficiency to 98% collection efficiency. This gives a cost of R0.55 per tonne of waste. This is in line with Morrisa et al. (2013) when compared to WC.

6.6.5.4 Leachate

While the digestate from the AD process can have high liquid content, it is assumed that this is used for land application as a liquid fertiliser. This is further confirmed by Morrisa et al. (2013) ranking of AD for eco toxicity below that of landfills and equal to composting.

6.6.5.5 Disamenity

Due to the lack of data on the disamenity of AD sites, similar assumptions to IVC will apply, essentially that the plant is located near the landfill and will have the same disamenity effect as compliant landfills with food waste diversion (R5.43 per tonne of food waste).

6.6.6 Total Benefit

The total benefit as seen in the summary of calculations in Table 50 is:

B_{AD} = - R4,752.32 million per year (status quo compliance)

B_{AD} = - R3,781.76 million per year (100% compliance)

Table 50: Summary of the economic assessment of impacts considered for anaerobic digestion per ZAR/year and ZAR/tonne (2022 ZAR), varying compliance.

Type of cost	Impact Group	Description of impact	Impact Identification		Impact Quantification (tonnes/year)	Impact Valuation	
			Cost (R/tonne /year)	Revenue (R/tonne /year)		Cost (Million R/year)	Revenue (Million R/year)
Internal	Infrastructure	AD O&M	1,153.75	-	3,030,271.60	3,496.18	-
Internal		Landfill O&M	484.67	-	33,254,208.23	16,117.17	-
Internal	Recovery of waste	Gate Fees – landfill	-	376.28	33,254,208.23	-	12,512.89
Internal		Gate fees – AD	-	376.28	3,030,271.60	-	1,140.23
Internal		Food waste to electricity	-	697.32	3,030,271.60	-	2,113.07
		Food waste to compost	-	551.28	3,030,271.60	-	1,670.53
External	Environmental	Chemical fertiliser replacement	-	46.63	3,030,271.60	-	141.30
External		Fossil fuel elec. displacement	-	167.86	3,030,271.60	-	508.66
External		GHG – landfill	57.75	-	33,254,208.23	1,920.57	-
External		GHG – AD	35.52	-	3,030,271.60	107.64	-
External		LAP – landfill	0.82	-	33,254,208.23	27.28	-
External		LAP – AD	0.55	-	3,030,271.60	1.67	-
External		Leachate – landfill	0.03	-	33,254,208.23	1.00	-
External		Leachate - AD	-	-	3,030,271.60	-	-
External	Quality of life	Disamenity - AD	5.43	-	3,030,271.60	16.45	-
External		Disamenity - landfill	34.61 (5.43)	-	33,254,208.23	1,151.05	-
Total of Internal Impacts						19,613.35	17,436.72
Total External Impacts						3,225.66	649.96
Total Impacts						**22,839.01**	**18,086.68**
Assuming 100% Compliance							
Total of Internal Impacts						19,613.35	17,436.72
Total External Impacts						2,255.10	649.96
Total Impacts						**21,868.45**	**18,086.68**

6.7 Vermicompost

6.7.1 Object Definition

The object of this scenario is to quantify the Total Benefit (B_T) of diverting food waste from 158 landfills in South Africa to produce Vermicompost (VC).

6.7.2 Project Impacts

The project impacts are listed in Table 51, quantification, and monetary evaluation of the impacts (private and external) for LFG collection systems will follow.

Table 51: Summary of the impacts of food waste diversion for Vermicomposting.

Type of cost	Impact Group	Description of impact	Impact Identification	
			Cost	Revenue
Internal	Infrastructure	Treatment and final disposal	OpEx and CapEx	
Internal	Recovery of waste	Gate Fees		Market Price
Internal		Compost Protein (feed)		Market Price
External	Use of materials	Landfill diversion		Avoid material sent to landfill
	Environmental	Replace chemical fertiliser		Reduces nitrogen fert. use + CO_2 sequestration + high protein feed
External		Emissions to air	GHG production	Reduction in landfill GHG
External		Emissions to air	LAP production	Reduction in landfill LAP
External		Emissions to ground	Leachate	Reduction in landfill leachate
External	Quality of life	Disamenities	Disamenity	

6.7.3 Infrastructure

As indicated previously, Table 13: Expected cost ranges for various AWT options in Chapter 2, details these costs from the literature and suggests an OpEx and CapEx cost for VC of R437 per tonne of organic waste processed.

Landfill private costs are assumed to stay the same as the status quo scenario, however the waste volumes processed will reduce as the food waste will be assumed to be diverted for alternative use.

6.7.4 Recovery from waste

6.7.4.1 Gate Fees

Gate fees landfilling: R376.28 per tonne waste. As per assumption for windrow composting.

6.7.4.2 By-Product: Vermicompost

Using the same process methodology as above for estimating by-product income. Lalander et al. (2015) find a 45% reduction in volume from manure based studies, however food waste has a higher water content, and the reduction can be between 40-60% reduction, this study will assume 60% as a conservative estimate. Current bulk retail prices for 30kg vermicompost is R137.00 (Organics, 2023) or R95.00 for 22kg (Farms, 2024) which equates to R5.31 to R4.32 per kg. This suggests a price of R1,728.00 per tonne of food waste, which is in line with the GreenCape (2022b) estimate of R1,980.00 per tonne of food waste. It is to be noted the considerable variation in compost price between different processes, this is due to the inherent quality of vermicompost vs traditional composting processes as well as biological benefits due to beneficial bacteria and archaea which promote plant growth (Yasmin et al., 2022).

6.7.4.3 By-Product: Protein

There is an additional by-product of this process, the worn biomass itself which can be used as a protein substitute for animal feed or sold for fish feed. There are no prices available for this, so a price is scaled from BSF to R25 per kg worm mass (similar price to bulk fishmeal), the worm biomass production is taken as 4.21 kg per tonne food waste (Yadav et al., 2015). This gives an estimated price of R105.25 per tonne of food waste.

6.7.5 External impacts

6.7.5.1 Reduced waste volume landfilled.

As discussed previously, 8% reduction in processed waste.

6.7.5.2 Greenhouse Gas

Based on the literature (Table 52) the average GHG emissions from the vermicomposting process is 27.48 kg CO_2 equivalent per tonne of waste. There is very little literature on food waste on its own and most studies refer to manure (which pushes up the CH_4 emissions), to mixtures of manures or high carbon organics and food waste (to get optimum water content) (Yasmin et al., 2022). The estimate of 27.48 kg CO_2 equivalent per tonne of waste is in line with Komakech et al. (2015) who compares AD, WC, VC and BSF technologies, both vermicompost and BSF processing have low process emissions when compared to AD and composting. This is discunted to R13.65 per tonne of waste.

Table 52: GWP of food waste from literature for Vermicomposting (kg CO2 equ/tonne waste).

Waste source	Process GWP	GWP fert. sub.	Total	Comment	Reference
Bio-waste	18.84	-1.1	17.7	LCA: Not including transport	(Komakech et al., 2015)
Manure	20.8	-3	17.8	Construction + process emissions	(Komakech et al., 2016)
Crop residue and manure	0.003-0.251			Not including pretreatment	(Yasmin et al., 2022)
House waste + cocnut fibre	0			CH_4 emissions negligible, not including pretreatment.	(Yasmin et al., 2022)
Home waste + mulches	4.32		4.32	CH_4 emissions negligible, not including pretreatment	(Chan et al., 2010)
Duck manure +reed straw and zeolite	15.08 – 53.79			CH_4 emissions negligible – mostly N_2O emissions	(Yasmin et al., 2022)
Sewage sludge +rice straw	3.6-211.43			CH_4 emissions negligible – mostly N_2O emissions	(Yasmin et al., 2022)
Cattle manure	159.6			Mostly from CH4	(Yasmin et al., 2022)
Vegetable, straw, manure	19.28-34	-45-68		50/50 N2o to CH4 – total without carbon	(Yasmin et al., 2022)
Average	42.93				
Average (no outliers)	**27.48**				

The GHG benefit due to fertiliser displacement is less clear from the literature, and the variations in the initial constituents of the waste stream have significantly varied

the potential results. Therefore, as the same input food waste is assumed to be used for all technologies, Table 40 above suggests 55.22 kg CO_2 eq/tonne of food waste. This is discounted to R27.42 per tonne of waste.

In addition to the above, another benefit to be considered is the replacement of protein feed with worm protein. Komakech et al. (2015) have suggested the substitution of silver cyprinid fish in animal feed with BSF and worm protein, and measured the avoided emissions based on reduction in road transport emissions (assuming fishing has zero emissions). For vermicompoting an estimated saving of 0.257 kg CO_2 eq/tonne of food waste, and 0.726 kg CO_2 eq/tonne of food waste for BSF. Scaling this against the benefit for BSF below gives R1.82 per tonne of food waste. Scaling by the worm biomass to BSF biomass gives R0.72 per tonne of food waste. This lower estimate will be used as a conservative estimate.

6.7.5.3 Local Air pollutants

There is no existing literature on the air pollutants produced by vermicomposting and any LCA only recognised the transport and preprocessing stages as having any potential impact (shredding and construction emissions) (Komakech et al., 2015). Since this study is not looking at transport emissions, and since there is such low energy use during the processing stage (no diesel used for turning or electricity for heating/aeration) (Yasmin et al., 2022), this study will assume the local air pollutants to be negligible.

6.7.5.4 Leachate

While worm farms do produce leachate from the digesting substrate (in a similar way to composting), which has the potential to contaminate surrounding environment, an optimal moisture content needs to be maintained throughout the production phase and it is assumed that the leachate is used in this process. Further any leachate produced is collected and sold as a soil amendment.

6.7.5.5 Disamenity

As discussed previously, any mismanaged food waste processing facility has the potential to attract pests, produce bad smells and have negative effects on the surrounding community remains a possibility. Further there is limited energy use in VC processing when compared to WC as it is not continuously turned by machinery.

Due to the lack of data on the disamenity of VC sites, similar assumptions to IVC and AD will apply, essentially that the plant is located near the landfill and will have the same disamenity effect as compliant landfills with food waste diversion (R5.43 per tonne of food waste).

6.7.6 Total Benefit

The total benefit as seen in the summary of calculations in Table 53 is:

B_{VC} = - R1,305.48 million per year (status quo compliance)

B_{VC} = - R 334.92 million per year (100% compliance)

Table 53: Summary of the economic assessment of impacts considered for vermicomposting per ZAR/year and ZAR/tonne (2022 ZAR), varying compliance.

Type of cost	Impact Group	Description of impact	Impact Identification		Impact Quantification (tonnes/year)	Impact Valuation	
			Cost (R/tonne /year)	Revenue (R/tonne /year)		Cost (Million R/year)	Revenue (Million R/year)
Internal	Infrastructure	VC O&M	437.00	-	3,030,271.60	1,324.23	-
Internal		Landfill O&M	484.67	-	33,254,208.23	16,117.17	-
Internal	Recovery of waste	Gate Fees – landfill	-	376.28	33,254,208.23	-	12,512.89
Internal		Gate fees – VC	-	376.28	3,030,271.60	-	1,140.23
Internal		Food waste to Feed	-	105.25	3,030,271.60	-	318.94
		Food waste to compost	-	1,728.00	3,030,271.60	-	5,236.31
External	Environmental	Chemical fertiliser replacement	-	27.42	3,030,271.60	-	83.09
External		Feed substitution		0.72	3,030,271.60	-	2.18
External		GHG – landfill	57.75	-	33,254,208.23	1,920.57	-
External		GHG – VC	13.65	-	3,030,271.60	41.36	-
External		LAP – landfill	0.82	-	33,254,208.23	27.28	-
External		LAP – VC	-	-	3,030,271.60	-	-
External		Leachate – landfill	0.03	-	33,254,208.23	1.00	-
External		Leachate - VC	-	-	3,030,271.60	-	-
External	Quality of life	Disamenity - VC	5.43	-	3,030,271.60	16.45	-
External		Disamenity - landfill	34.61 (5.43)	-	33,254,208.23	1,151.05	-
Total of Internal Impacts						17,441.40	19,208.37
Total External Impacts						3,157.72	85.27
Total Impacts						**20,599.12**	**19,293.64**
Assuming 100% Compliance							
Total of Internal Impacts						17,441.40	19,208.37
Total External Impacts						2,187.16	85.27
Total Impacts						**19,628.56**	**19,293.64**

6.8 Black soldier fly processing

6.8.1 Object Definition

The object of this scenario is to quantify the Total Benefit (B_T) of diverting food waste from 158 landfills in South Africa to produce compost and protein using black soldier fly larvae (BSF).

6.8.2 Project Impacts

The project impacts are listed in Table 54, quantification, and monetary evaluation of the impacts (private and external) for LFG collection systems will follow.

Table 54: Summary of the impacts of food waste diversion to BSF processing facility.

Type of cost	Impact Group	Description of impact	Impact Identification	
			Cost	Revenue
Internal	Infrastructure	Treatment and final disposal	OpEx and CapEx	
Internal	Recovery of waste	Gate Fees		Market Price
Internal		Compost Protein (feed)		Market Price
External	Use of materials	Landfill diversion		Avoid material sent to landfill
	Environmental	Replace chemical fertiliser		Reduces nitrogen fert. use + CO_2 sequestration
External		Emissions to air	GHG production	Reduction in landfill GHG
External		Emissions to air	LAP production	Reduction in landfill LAP
External		Emissions to ground	Leachate	Reduction in landfill leachate
External	Quality of life	Disamenities	Disamenity	

6.8.3 Infrastructure

As indicated previously, Table 13: Expected cost ranges for various AWT options in Chapter 2, details these costs from the literature and suggests an OpEx and CapEx cost for BSF facility is R661.8 per tonne of organic waste processed.

6.8.4 Recovery from waste

6.8.4.1 Gate Fees

Gate fees landfilling: R376.28 per tonne waste. As per assumption for windrow composting.

6.8.4.2 By-Product: Protein, Oil, Frass (compost)

Based on the literature on food waste processing with BSF, there is considerable variation in the potential output of this process. Ites et al. (2020) emphasise this by comparing three different waste streams; spent brewery grains, potato peels and expired food waste which produced 28, 32 and 9.6kg per tonne waste of dried larvae and 120, 112 and 112kg per tonne waste of frass respectively. Further literature for food waste suggest 1 tonne food waste produces; 30.94kg dried larve and 300kg matured compost (Guo et al., 2021), and 30kg dried larve and 335kg matured compost (Salomone et al., 2017). Nyakeri et al. (2017) suggest the bioconversion rate can vary from 11.8% to 20.8% and a substrate reduction of 68% to 81.8%, with the higher values attributed to kitechen food waste.

Additional high-value byproducts that can potentially be produced is the oil and high protein cake, however since these require additional processing steps, with their own associated GHG footprint, these will not be considered. It is to be noted however that this can considerably increase the value derived from the BSF output.

Based on the literature, this study will assume the following:

1 tonne food waste produces 30kg dried larvae and 300kg frass (as matured compost).

A review of the market prices (2022) for these by-products from the largest producers and wholesalers is South Africa

- Live larvae: R2,875.00 per 200kg, equates to R14.38 per kg (Nambu, 2022)

- Frass: R15 per kg (Insecto, 2022) or R2000/tonne (R2/kg) (Maltento, 2023; BSF Breeding, 2024)
- Dried Larvae: R45/kg (defatted) + R45/kg oil = R60.90 (Insecto, 2022), R65/kg (Maltento, 2023)

This suggests the value of BSF frass and dried larvae to be between R2,427.00 and R6,450.00. This can vary considerably but is in line with the GreenCape (2022b) which proposes value added potential for BSF between R2,028.00 to R10,170.00 per tonne organic waste. This does not include the high-value products that can be drives from BSF.

6.8.5 External impacts

6.8.5.1 Reduced waste volume landfilled.

As discussed previously, 8% reduction in processed waste.

6.8.5.2 Greenhouse Gas

Study by Mertenat et al. (2019) suggests that BSF treatment emits 47 times less than WC. Table 55 summarises the GWP from the literature based on LCA of the BSF system, the variation in figures is attributed to the different assumptions in system boundaries, whether construction emissions were accounted for and the energy source (high carbon or renewable mix) (Boakye-Yiadom et al., 2022). Taking an average of results (using sources that have similar energy mixes to SA), gives 60 kg CO_2 eq/tonne of food waste for the process emissions, when discounted this is estimated to be R29.80.

GHG benefit for the substitution of chemical fertiliser from the literature varies significantly between 1.5 and 486.08 with an average of 160 kg CO_2 eq/tonne of food waste. This is due to varying assumptions, in order to keep all assumptions constant, this study will use 49.71 kg CO_2 eq/tonne of food waste from Table 40 which is discounted to R24.69 per tonne of waste.

The GHG benefit from substituting animal feed with BSF larvae is especially dependent on assumptions and the feed chosen to be replaced. Table 55 summarises the literature and the average benefit is taken to be 10.34 kg CO_2 eq/tonne of food waste, which is discounted to R5.14 per tonne.

Table 55: GWP of food waste from literature for BSF processing (kg CO2 equ/tonne waste).

Waste source	Process GWP	GWP fert. sub.	GWP feed sub.	Total	Comment	Reference
Bio-waste	13.89	-0.84	-0.73	12.40	LCA: Not including transport, constr.	(Komakech et al., 2015)
Food Waste	90.05 (17.24 emissions)	-52.69	-20	17.36	LCA: Not including transport, constr.	(Guo et al., 2021)
Kitchen waste	50 (post composting 32)	-	-15	35	LCA: Process (no transport) + post composting – fishmeal production and transport (substitution)	(Mertenat et al., 2019)
Food waste + pruning waste	24.8	-426.38	-5.62	-432	LCA: transport removed, (elec from low carbon sources)	(Salomone et al., 2017)
Food waste	67.47	-487.08 (combined benefit)		-419.61	LCA: removed transport and collection (5%)	(Mondello et al., 2017)
Food waste Brewery grain Potato peels				19.10 -182.59 252.08	LCA: including construction emissions and transport	(Ites et al., 2020)
Vegetable waste	66.67	-	-	-	LCA: Not including transport, constr. Or substitution	(Nugroho et al., 2023)
Food waste	140.91	-		-	LCA: including const., transport	(Pahmeyer et al., 2022)
Average	64.82	-159.97	-10.34	-87.28		
Average (no outliers)	**60.00**		**-10.34**			

6.8.5.3 Local Air pollutants

There is limited literature on the air pollutants produced for BSF and any LCA only recognises the transport and preprocessing stages as having any potential impact (Komakech et al., 2015). Energy use is similar to that of VC (Komakech et al., 2015) and when compared to WC, BSF has increased environmental benefits in every category (Mondello et al., 2017). Since this study is not looking at transport emissions, and since there is such low energy use during the processing stage. This study will assume the local air pollutants to be negligible.

6.8.5.4 Leachate

While BSF does produce leachate from the digesting substrate (in a similar way to composting), which has the potential to contaminate surrounding environment, an optimal moisture content needs to be maintained throughout the production phase and it is assumed that the leachate is used in this process.

6.8.5.5 Disamenity

As discussed previously, any mismanaged food waste processing facility has the potential to attract pests, produce bad smells and have negative effects on the surrounding community remains a possibility. Further there is limited energy use in VC processing when compared to WC as it is not continuously turned by machinery.

Due to the lack of data on the disamenity of VC sites, similar assumptions to IVC and AD will apply, essentially that the plant is located near the landfill and will have the same disamenity effect as compliant landfills with food waste diversion (R5.43 per tonne of food waste).

6.8.6 Total Benefit

The total benefit as seen in the summary of calculations in Table 56 is:

B_{BSF} = - R 231.28 million per year (status quo compliance)

B_{BSF} = R 739.28 million per year (100% compliance)

Table 56: Summary of the economic assessment of impacts considered for BSF per ZAR/year and ZAR/tonne (2022 ZAR), varying compliance.

Type of cost	Impact Group	Description of impact	Impact Identification		Impact Quantification (tonnes/year)	Impact Valuation	
			Cost (R/tonne /year)	Revenue (R/tonne /year)		Cost (Million R/year)	Revenue (Million R/year)
Internal	Infrastructure	BSF O&M	661.80	-	3,030,271.60	2,005.43	-
Internal		Landfill O&M	484.67	-	33,254,208.23	16,117.17	-
Internal	Recovery of waste	Gate Fees – landfill	-	376.28	33,254,208.23	-	12,512.89
Internal		Gate fees – BSF	-	376.28	3,030,271.60	-	1,140.23
Internal		Food waste to high protein feed	-	1,827.00	3,030,271.60	-	5,536.31
		Food waste to compost	-	600.00	3,030,271.60	-	1,818.16
External	Environmental	Chemical fertiliser replacement	-	24.69	3,030,271.60	-	74.82
External		Feed substitution	-	5.14	3,030,271.60	-	15.58
External		GHG – landfill	57.75	-	33,254,208.23	1,920.57	-
External		GHG – BSF	29.80	-	3,030,271.60	90.30	-
External		LAP – landfill	0.82	-	33,254,208.23	27.28	-
External		LAP – BSF	-	-	3,030,271.60	-	-
External		Leachate – landfill	0.03	-	33,254,208.23	1.00	-
External		Leachate - BSF	-	-	3,030,271.60	-	-
External	Quality of life	Disamenity - BSF	5.43	-	3,030,271.60	16.45	-
External		Disamenity - landfill	34.61 (5.43)	-	33,254,208.23	1,151.05	-
Total of Internal Impacts						18,122.60	21,007.59
Total External Impacts						3,206.66	90.39
Total Impacts						**21,329.26**	**21,097.99**
Assuming 100% Compliance							
Total of Internal Impacts						18,122.60	21,007.59
Total External Impacts						2,236.10	90.39
Total Impacts						**20,358.70**	**21,097.99**

CHAPTER 7: RESULTS

7.1 Landfilling in South Africa

7.1.1 Status quo of landfilling in South Africa

The private costs to landfill in South Africa are made up of the capital, operational costs and gate fees. In this study the operational and capital costs are an average across the country per tonne of waste, however this may vary and a more rigorous analysis on costs for a South African context may find costs vary depending on location, size of landfill and climate.

A review of the landfill gate fees across South Africa indicates that there is considerable variation between provinces as well as municipalities (see Table 13). In general, MSW, garden refuse and builders' rubble have the lowest gate fees with higher rates being applied for disposal of putrescible waste, this is because of the additional work required to ensure safe disposal as well as the external impact on the landfill. Private companies who need to safely dispose of this waste need to ensure that it cannot be recirculated to the public where it can potentially sicken someone or cause harm. The landfills therefore usually bury this waste deep in the landfill body to ensure it cannot be accessed, hence the additional costs. There is also an increase in leachate, odour and GHG production because of the higher moisture and carbon content in the putrescible waste which means the management and treatment costs increase.

Considering that financial viability is defined as the $B_P>0$, total private income should cover the private cost to landfill, only the Western Cape, Kwa-Zulu Natal and the Free State cover their expenses with gate fees, as seen in Figure 10. The gate fees for the Western Cape and Kwa-Zulu Natal are expected to cover the current capital, operational and close out cost for landfilling. The Western Cape has the highest gate fees in SA and it is expected to rise above inflation in the coming years (CoCT, 2021a). This should motivate waste generators and service providers to seek cheaper alternatives to landfilling.

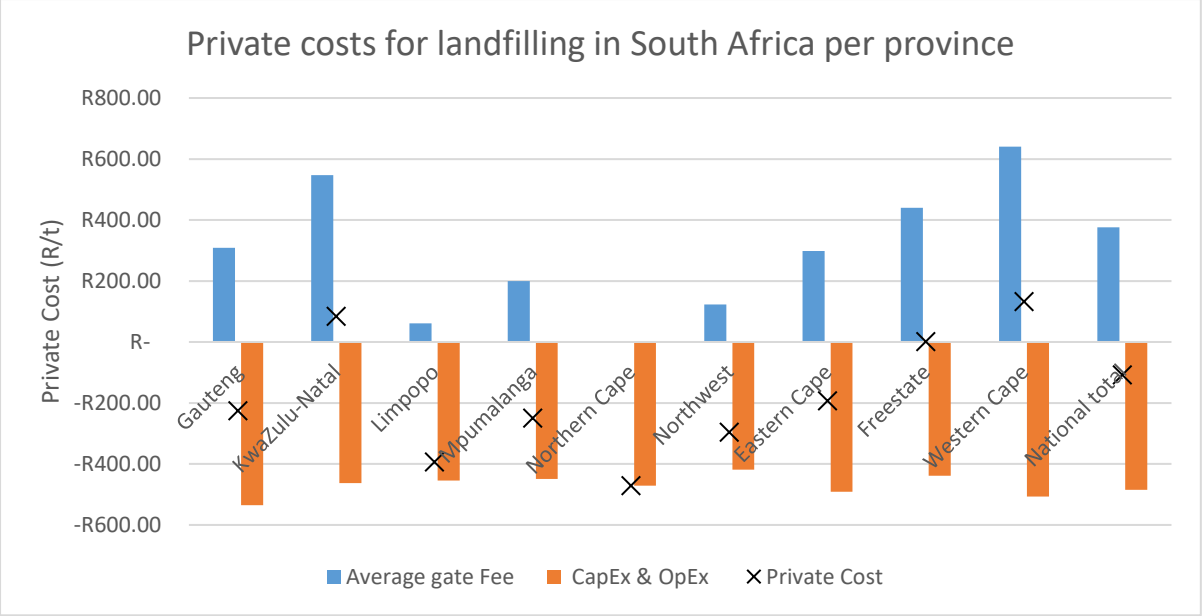

Figure 10: Estimated landfill gate fees and CapEx and OpEx costs per province in SA

All other provinces have average gate fees below the cost of landfilling, which immediately suggests that the current status quo is not financially viable. Of particular interest is the Northern Cape which does not charge any gate fees and has no compliant landfills. The Free State, which has high disposal fees but no compliant landfills, suggests serious mismanagement of funds. Compliance ben be seen in Figure 11.

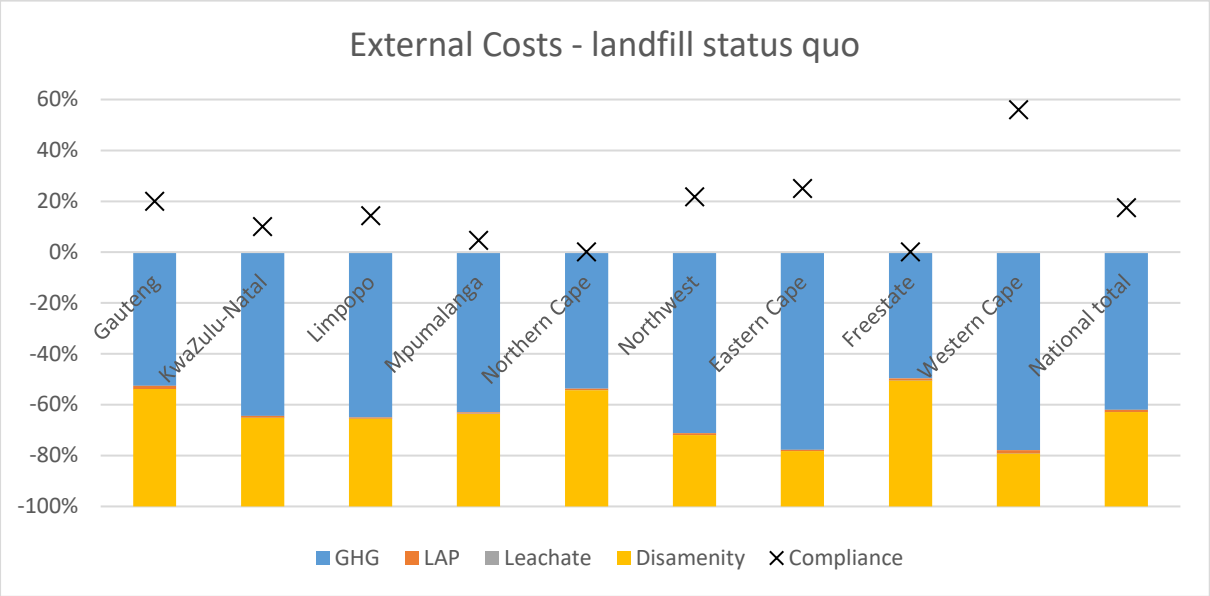

Figure 11: Proportion of external costs per province for the status quo of landfilling in SA.

The total cost of landfilling is South Africa as determined in section 6.1 Landfilling (status quo), estimates the cost of landfilling to be -R 8,753.18 million per year. The negative value indicated that there is a significant total cost to landfilling. This is due to the external costs associated with GHG emissions, leachate, LAP and disamenity costs. In fact, the low compliance rates further impact the disamenity cost, as seen in Figure 11, the lower the compliance the higher the disamenity cost is as a proportion of the total external cost.

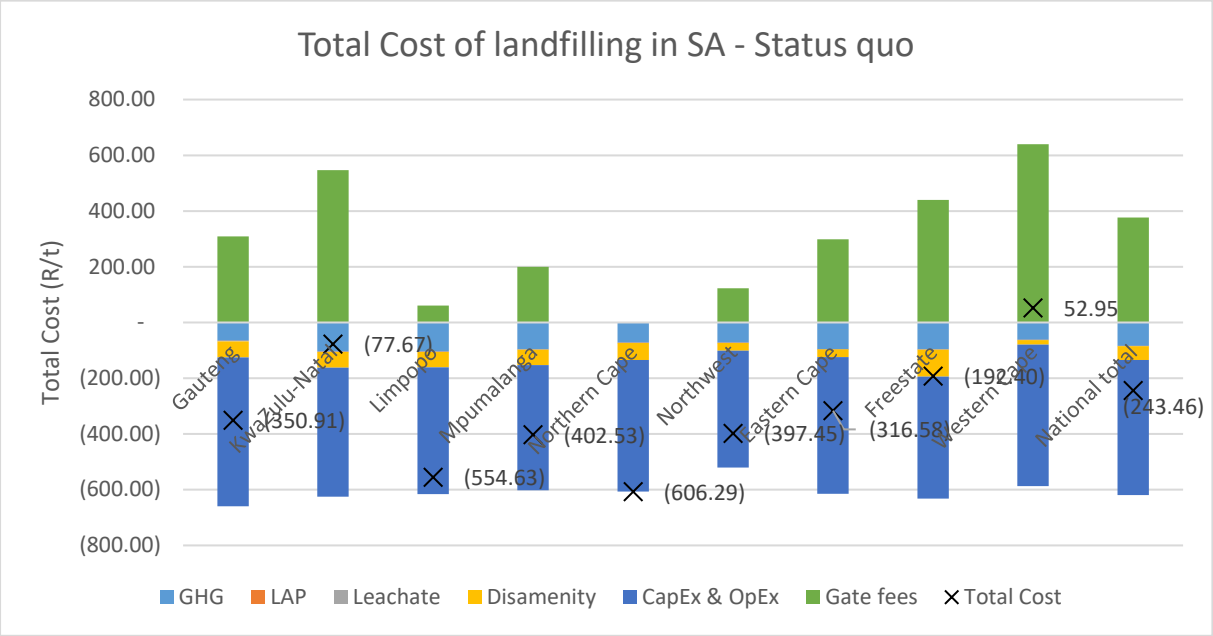

Figure 12: Internal, external and total status quo cost of landfilling in SA per province.

When comparing the total cost per province, as seen in Figure 12, only the Western Cape has a positive total benefit. This suggests that current landfill management system is economically, financially, socially, and environmentally viable. All other provinces have a negative total benefit with Limpopo and Northern Cape having the worst total benefit.

7.1.2 Effects of compliance

The compliance of the 158 landfills has a direct impact on the disamenity of these landfills. With the current status quo the disamenity ranges from 50% to 20% of the total external cost to landfilling (Figure 11). The Free State has a positive private benefit, which would suggest it is financially viable, however due to the low adherence to minimum landfill best practice, the high disamenity cost to surrounding

communities as well as other externalities clearly indicates that landfilling in the Free State comes at a significant cost to the fiscus and surrounding communities. When assuming all landfills in South Africa meet the minimum 80% compliance level, the disamenity cost reduces considerably and GHG emissions form the bulk of the external cost as seen in Figure 13.

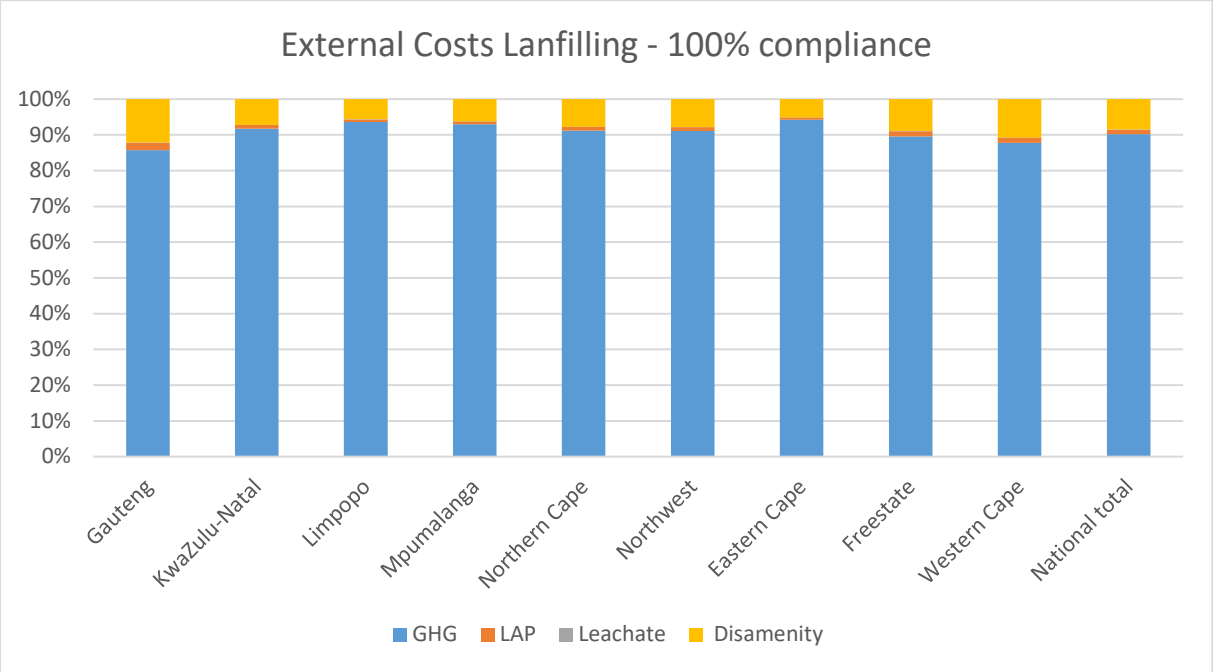

Figure 13: Proportion of external costs for 100% compliant landfills in SA.

The final cost with fully compliant landfills in South Africa is - R8,753.18 million per year as seen in in section 6.1, and the effect of making all landfills compliant saves R1,534.45 million in disamenity costs per year. That is as much as a 18% reduction in total external costs and an 84% reduction in the disamenity cost across South Africa.

7.2 Technology comparison

7.2.1 Private Benefit

The private benefit for each technology is made up of the capex and opex, gate fees and opportunity cost of any recovered benefit from diverted food waste. When comparing each technology proposed for food waste diversion both VC and BSF provide the highest potential benefit, followed by AD and WC. Only IVC has a $B_P<0$ which suggests it is not operationally and economically viable, as seen in Figure 14.

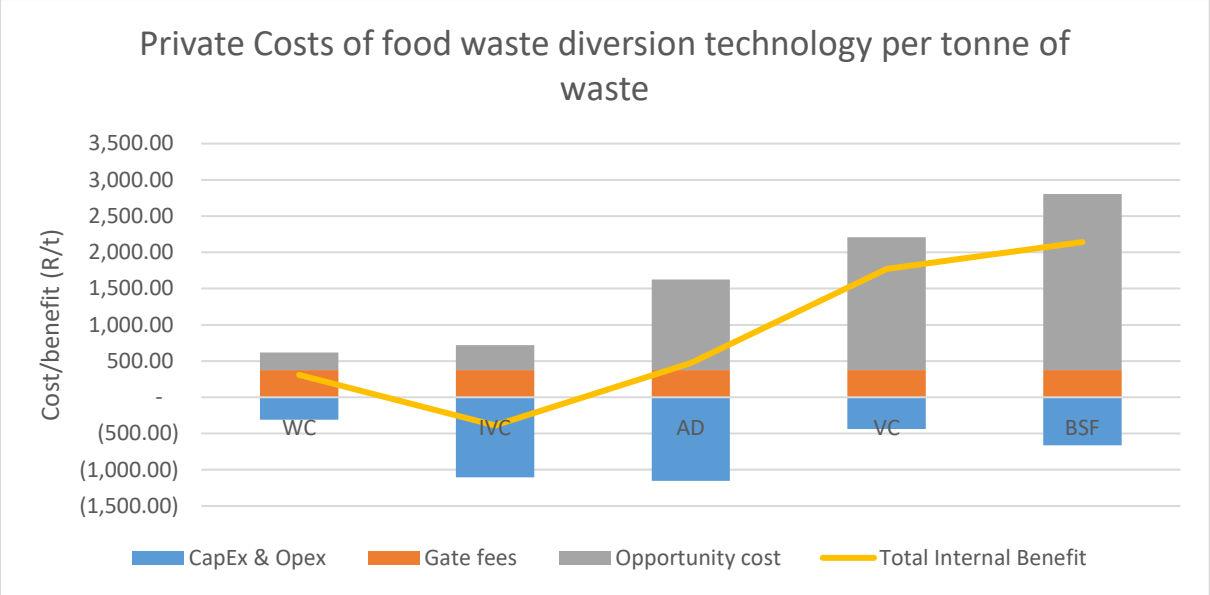

Figure 14: Summarising the private costs and benefits of food waste diversion technology and their potential financial viability (per tonne of food waste).

The impact of opportunity costs on the private cost is significant as it makes up the bulk of the private benefit for VC and BSF, 71% and 65% respectively. This is due to the high market price of VC and BSF byproducts. Of the diversion technologies, the ranking of the total benefits is as follows: BSF, VC, AD, WC and finally IVC with a negative. Note this ranking only looks at the diverted food waste and does not account for any benefits of diverting the waste stream from landfill.

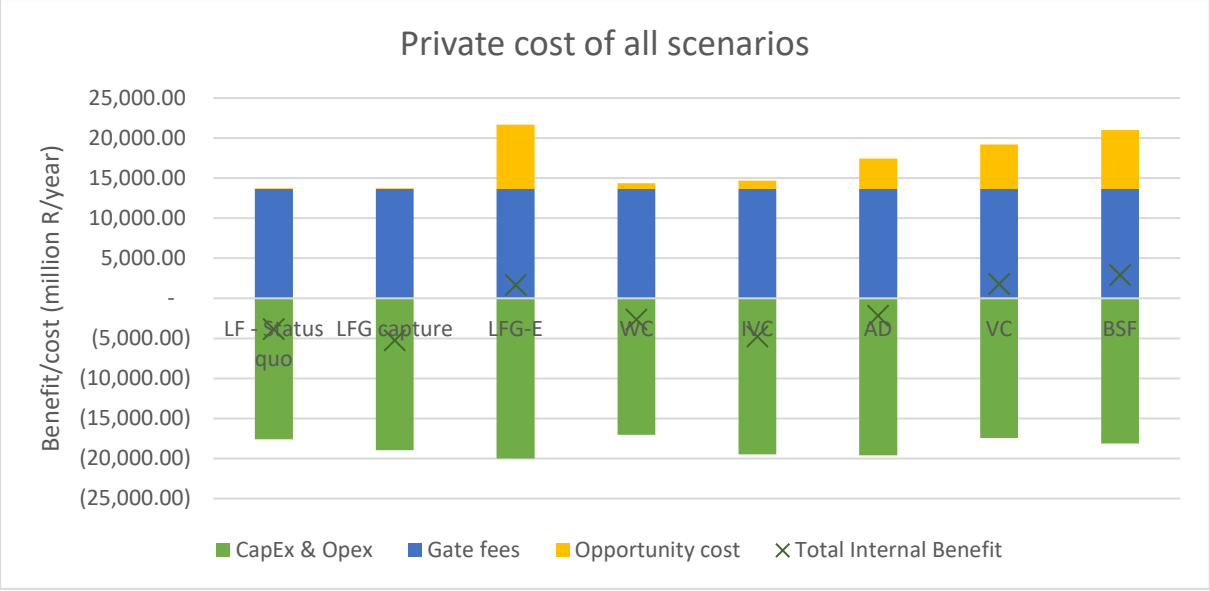

Figure 15: Summary of private costs applied to entire annual waste volumes for South Africa for each scenario.

To compare the private benefit of all scenarios, the costing needs to be applied to the volume of waste being processed, this is summarised in Figure 15. This includes the cost of landfilling the remaining waste once food waste has been diverted. Economically viable options, where $B_P>0$, includes BSF, VC and LFG-E, with AD and WC falling in the negative mainly due to the negative cost of landfilling. Importantly LFG capture and flaring has a lower B_P than the landfilling status quo, it is to be noted that the minimum national requirement (DWAF, 1998) requires landfills over a certain size to manage the GHG emissions due to their impacts on the surrounding communities and high global warming impacts. The status quo B_P are then expected to increase from their current state.

7.2.2 External Benefit

The external benefit, described previously as the sum of the external revenue minus the sum of the external costs, is summarised in Figure 16. It can be seen that the environmental benefit of anaerobic digestion outperforms all technology options, followed by VC, BSF and WC. All these options provide higher external benefits than all other landfilling options, including LFG-E, except IVC which has lower external benefits than LFG-E.

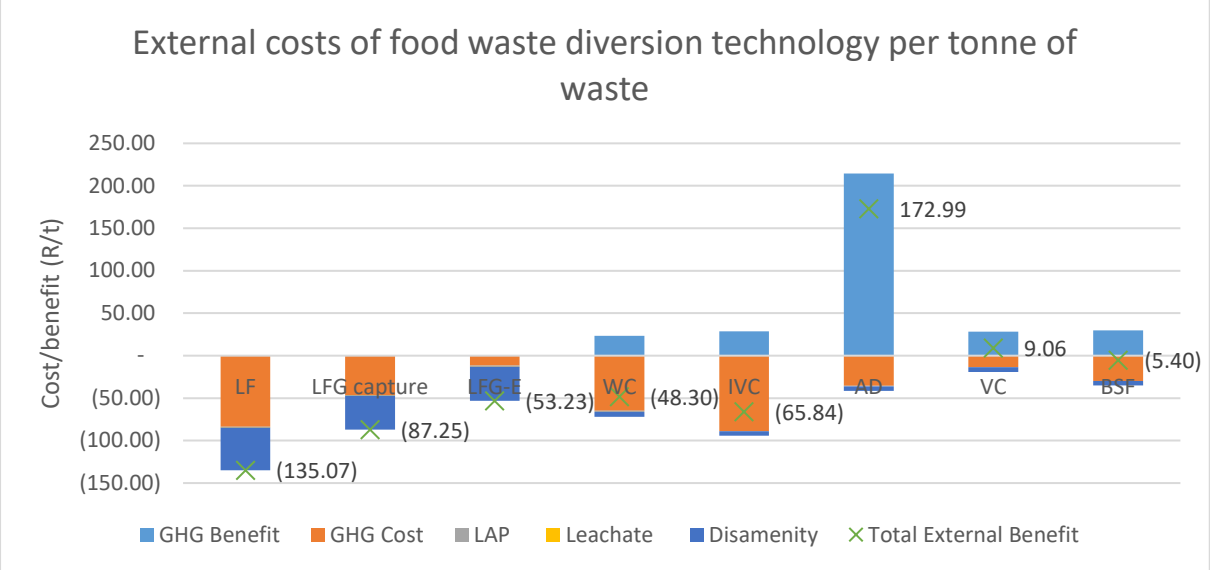

Figure 16: Summarising the external costs and benefits of food waste diversion technology (per tonne of food waste).

Applying the external costs to the total waste volumes is summarised in Figure 17 and Figure 18. These are for two scenarios, the current status quo with a high landfill

non-compliance and assuming full compliance of landfills nationally which has a reduced disamenity cost.

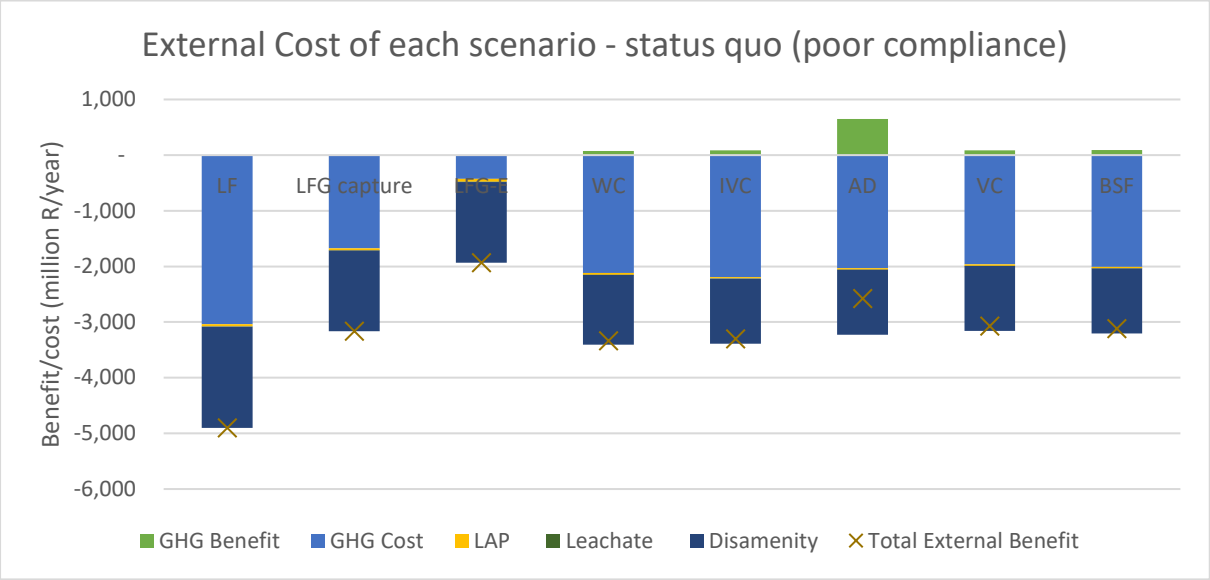

Figure 17: Summary of external benefit for each scenario assuming current state of compliance.

Figure 17 indicates that the current status quo landfilling has the worst external benefit, followed by WC and IVC, this would be due to the high disamenity cost as well as GHG emissions from the aerobic processed of WC and IVC. LFG capture and flaring has the same cost as BSF which is closely followed by VC, with the highest external benefit going to LFG to energy and AD. This is due to the displacement of GHG in both scenarios which is assumed to displace electricity from coal fired power stations. Note that for LFG-E, the GHG cost and benefit are already combined and not indicated separately. Interestingly, the GHG emission from landfill still makes up 66-85% of the external cost due to the remaining organic fraction in the landfill, this suggests a combination of scenarios may significantly reduce the GHG emissions and make other options more viable.

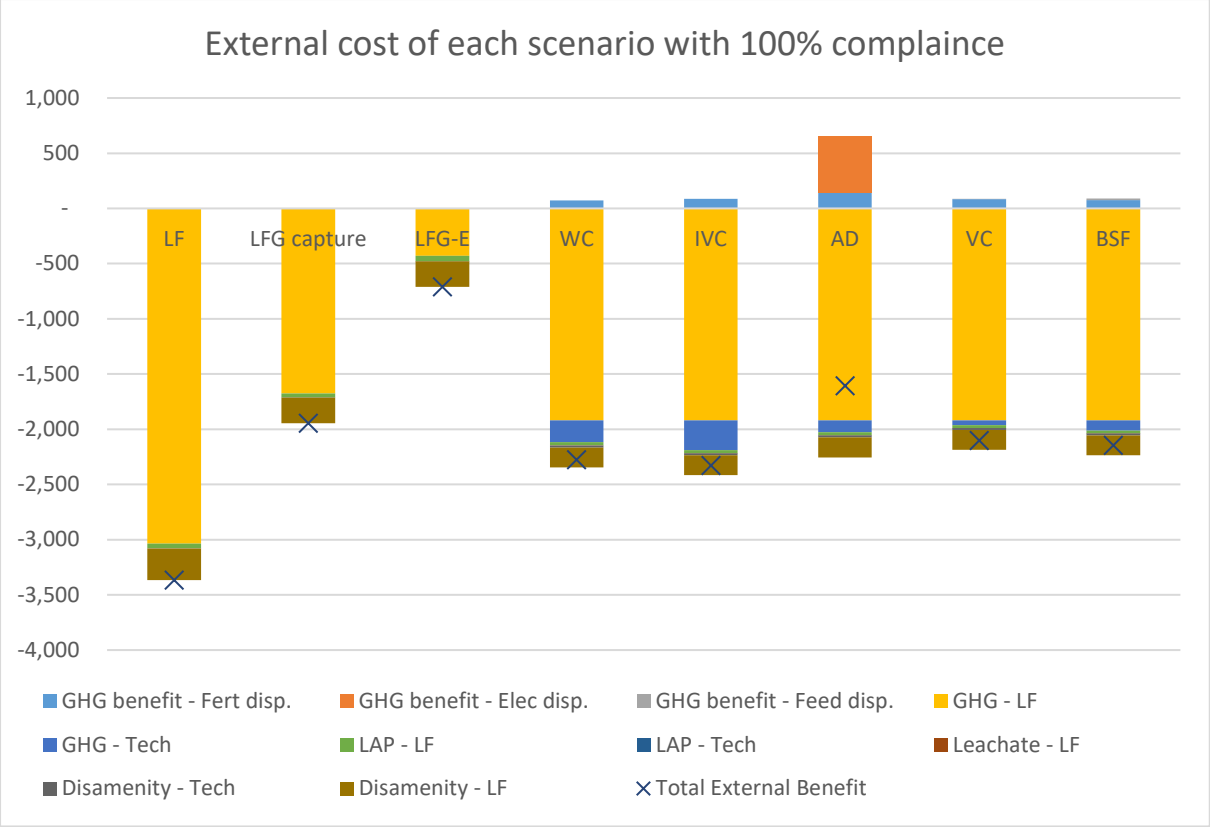

Figure 18: Summary of external benefit for each scenario assuming 100% compliance across all sites.

This is clearer in Figure 18, which clearly shows the impact of GHG emissions of food waste diversion technologies being heavily dominated by the LF GHG emissions. With the scenario of 100% compliant landfills (Figure 18), LFG to energy is the preferred option followed by AD and similar ranking between the technologies as discussed above.

7.2.3 Total Benefit

The total benefit, defined as the sum of the private and external benefits, gives an indication which choice of scenario will give the greatest benefit. This varies depending on the compliance of the landfill sites as that impacts the disamenity cost. On a per tonne of food waste basis, BSF and then VC have the highest benefit, followed by AD and then WC (Figure 19). All technologies have a total benefit, except for IVC which has a negative benefit. Again, the above technologies all provide a total benefit over all three landfilling options on a per tonne of waste basis.

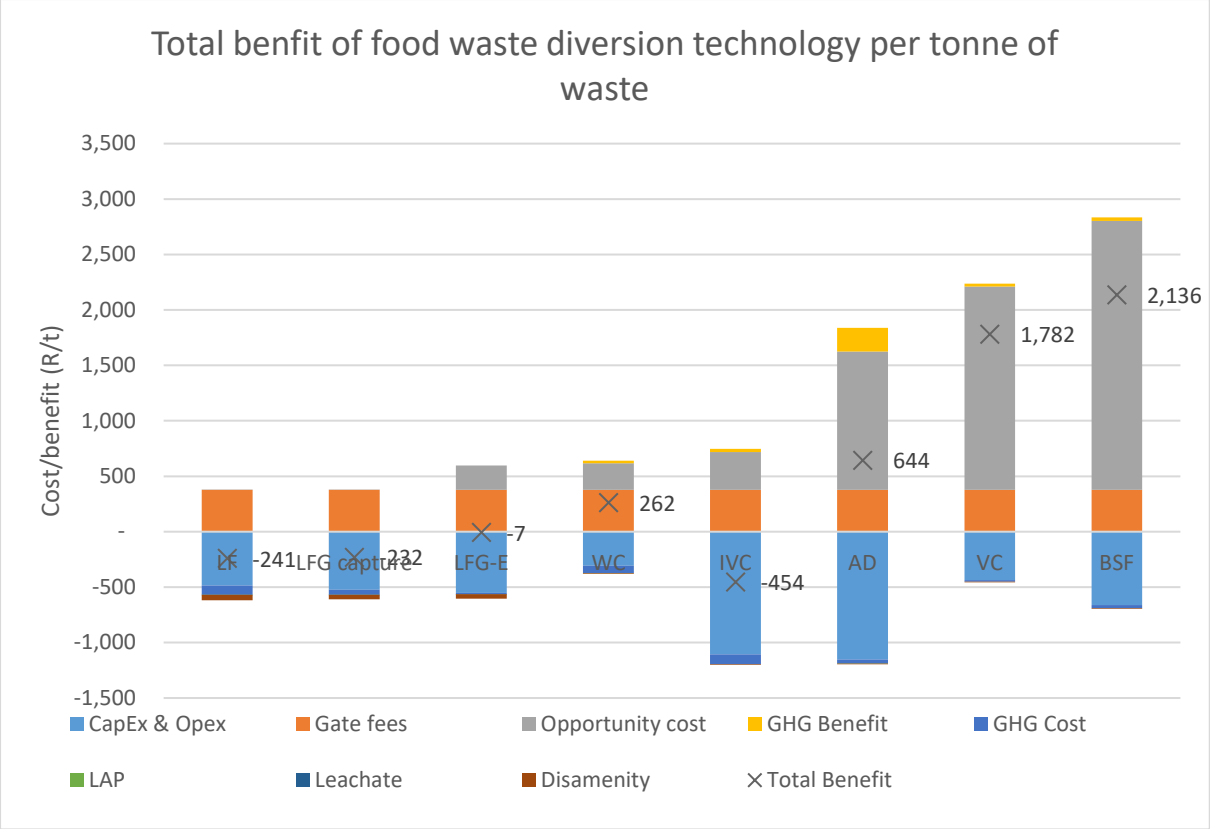

Figure 19: Summarising the total costs and benefits of food waste diversion technology (per tonne of food waste).

As seen in Figure 20 and Figure 21, the ranking of scenarios is affected by compliance, with highest B_T (for current compliance levels) going to BSF followed by LFG-E, VC, AD, WC. The status quo landfilling has the lowest B_T, with LFG capture and flaring reducing the GHG emissions and disamenity costs having a better total benefit. Since IVC has a $B_P<0$ it is disqualified from the analysis as not being financially viable in South Africa.

If all landfills were to meet minimum compliance levels, the ranking changes, and LFG-E has the highest B_T, followed by BSF, VC, AD, and WC. Compliant LF and LFG capture and flaring have a similar B_T; interestingly, the increased capital costs for flaring are offset by the GHG reduction benefits.

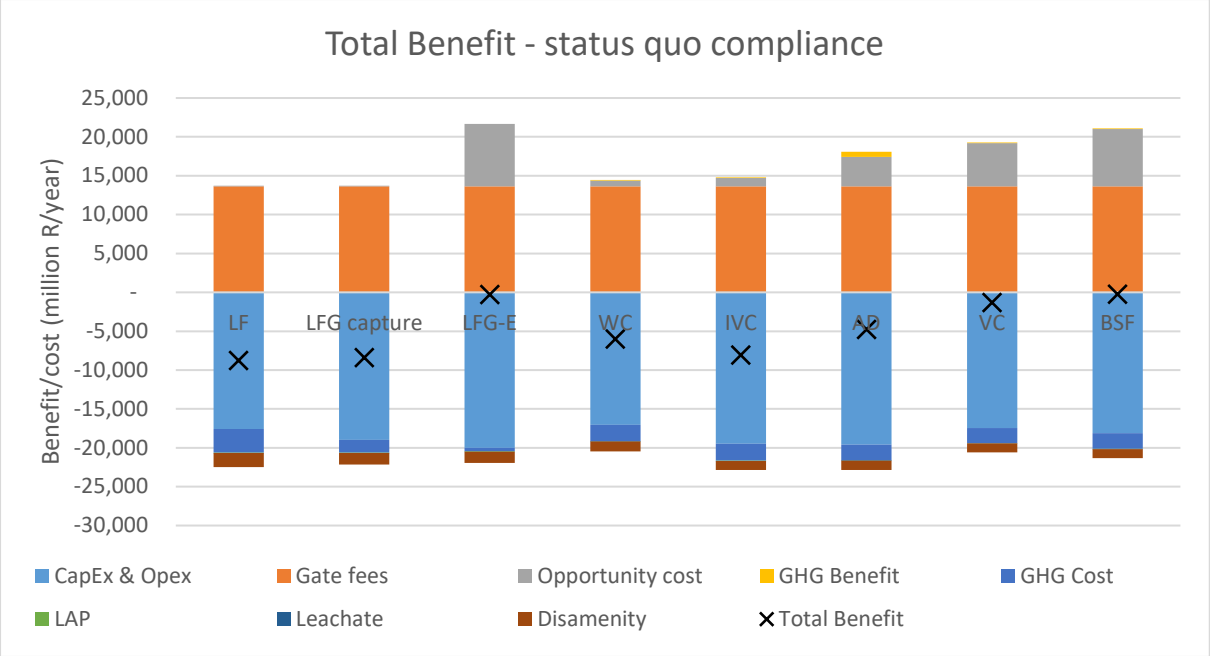

Figure 20: Summarizing the total benefit for each scenario - assuming status quo landfill compliance.

Figure 20 and Figure 21 also show the breakdown of B_T which makes clear the impact of gate fees and capital and operational costs. These make up much of the B_T, with 43-53% for CapEx and OpEx, and 31-38% for gate fees. This is followed by the opportunity benefit derived from food waste which ranges from 2-3% for WC and IVC, to 09-18% for energy benefit and BSF products. The GHG emission for each scenario is highest for LF at 9% and WC at 6% of total B_T, and ranges between 1-5% of B_T for the other scenarios.

The installation of LFG-E at all medium and large general waste landfills across South Africa would in theory provide an additional total benefit of R8,481 million per year for non-compliant landfills, and R8,167 million per year for higher landfill compliance levels. Similarly, the diversion of food waste to BSF processing facilities, would see an increase in total benefit of R8,522 million per year and R7,958 million per year for varying compliance.

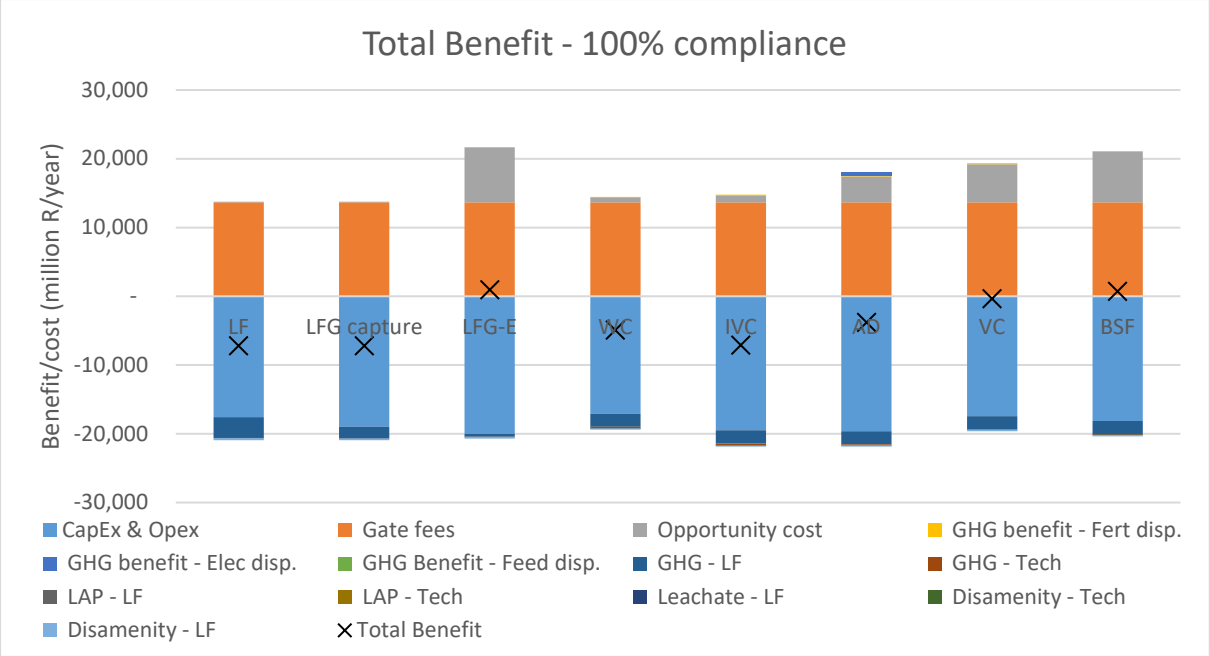

Figure 21: Summarising the total benefit for each scenario - assuming 100% landfill compliance.

The total benefits for status quo compliance and assumed 100% compliance can be seen in Figure 22. These will be further interrogated with the sensitivity analysis.

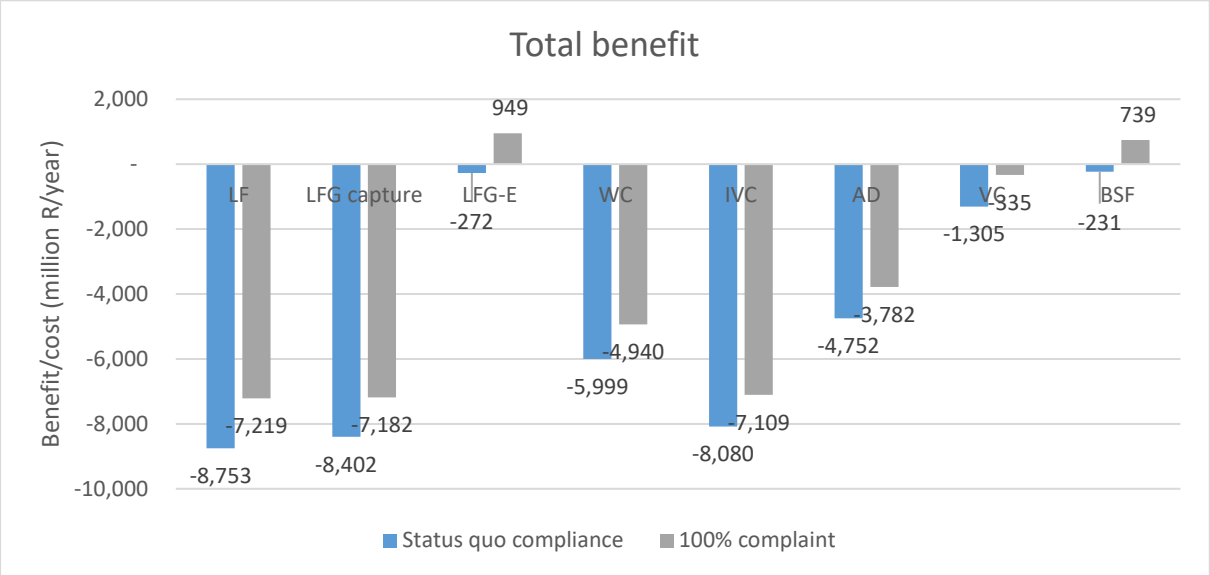

Figure 22: Graph summarising the total benefit for all scenarios for varying compliance levels.

7.3 Sensitivity analysis

Based on Figure 21, the items that have the largest impact on the total benefit are capex and opex costs, gate fees, opportunity costs and GHG emissions. This report will look at varying the gate fees and base assumptions on LFG-E sites.

7.3.1 LFG to energy assumptions

The base assumption for LFG to energy scenario is that only landfills of a certain size are feasible for this technology. This assumption then only applies to 65 of the 158 landfills which are either medium or large, however a full feasibility study will need to be done before the viability, volume and quality of LFG is determined to be able to produce sufficient electricity for the investment. It is possible that only large landfills are viable options for LFG to energy projects, with this assumption only 17 of the 158 landfills are eligible for LGF to energy installations.

However, the assumption that only large landfills are viable for LFG-to-energy should not exclude medium sized landfills from LFG capture and flaring. The management of GHG in medium sized landfills is still a requirement (DWAF, 1998), and it can further be assumed that medium landfills have LFG capture installed. The sensitivity analysis will therefore look at 3 assumptions: LFG-to-energy for medium and large landfills (LFG-E L&M), LFG-to-energy on large landfills only with medium and small remaining as is (LFG-E Lo), and LFG-to-energy on large landfills with LFG capture on medium landfills (LFG-E LMc).

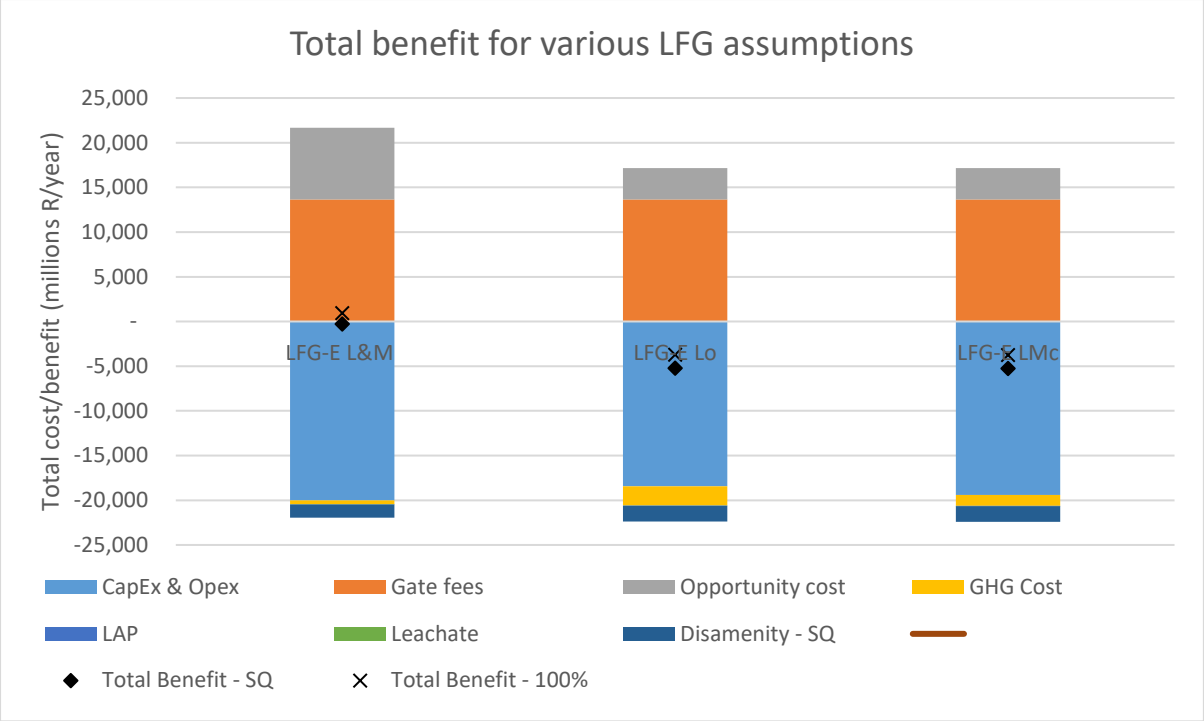

Figure 23: Summary of the sensitivity analysis to LFG-to-energy applied to medium and large landfills or only large with varying compliance levels.

The results of the above three scenarios are summarised in Figure 23, where the highest benefit is LFG-E L&M. This is mainly due to the large opportunity cost of electricity production which makes up for the higher CapEx and OpEx costs as well as the reduced GHG emissions. There is a significant increase in GHG emission cost for LFG-E Lo and LFG-E LMc, 4.9 times and 2.8 times increase respectively. This has an overall impact on the B_T from potentially being viable if implemented in large and medium landfills to a significant drop in viability (B_T<0) of 19.3 times for LFG-E Lo and 19.4 times drop for LFG-E LMc. Both LFG-E Lo and LFG-E LMc have B_T similar in magnitude. Due to the impact of this assumption on B_T any result when comparing technologies needs to be critically analysed due to the significant variance in potential benefit.

7.3.2 Gate Fees

The weighted gate fee has a significant impact on the results and makes up 30-45% of the total benefit. Due to the variance in fees across the country, municipalities and even landfill sites themselves a sensitivity analysis will assist in understanding the impact and importance of these fees. The lowest and highest landfill gate fees are in

the Northern Province (R0.00) and Western Cape (R640.00) respectively: both will be included in the analysis.

The weighted average of gate fees required to make all landfills, landfills with LFG capture and LFG-E financially viable (B_P>0) is R481.06, R520.59 and R330.54 per tonne of waste, respectively. Interestingly the B_P for LFG-E is lower than the current status quo gate fee, which is due to the projected electricity production which will generate a revenue. The worst case is considered under the assumption that national policy requires landfills to manage methane production and at a bare minimum LFG capture and mitigation is a requirement for medium and large landfills (DWAF, 1998).

The gate fees that are required to ensure B_T >0 will be significantly higher due to the addition of external costs and for landfilling and LFG capture and flaring this is R616.13 and R607.84 per tonne of waste, respectively. The required gate fee for LFG-E is not considered because in the case if 100% compliant landfills, there is a positive benefit which suggests the current gate fees are sufficient.

For the diversion technologies it was assumed that the gate fees remained the same as the municipal landfill fees, the assumption being that there is still a sorting and processing cost which may take place at the landfill. However, this may not be the case especially considering the benefit of diverting waste to landfill, municipalities would encourage the diversion of food waste by reducing these fees or having no fees. In fact, current municipal garden waste facilities do not charge feed for organic garden refuse. In most cases analysed, the B_p >0 with only IVC and WC having a B_P<0, in the case of WC by a small margin, and the gate fee required to make B_T>0 for WC is R114.52.

Based on this and summarised in Table 57, are the 4 scenarios that will be analysed in the sensitivity analysis, this is: GF-1 which assumes no fees for any waste management as is the case in the Northern Cape, GF-2 which looks at breakeven point (B_P>0) for landfill options and a lower gate fee of R66.22 for diversion (breakeven of WC), GF-3 which assumes that highest fees (as in the Western Cape) which cover costs for LFG management as well as post closure costs compared to a

higher gate fee for diversion covering total costs of WC, and finally, GF-4 which uses the weighted status quo gate fees and assumed no fee for diverted food waste.

Table 57: Variation in gate fees for the sensitivity analysis.

Technology	Gate fee (R/tonne of waste)			
	GF-1	GF-2	GF-3	GF-4
LF, LFG capture, LFG-E	0	520.59	640	376.28
WC, IVC, AD, VC, BSF	0	66.22	114.52	0

Figure 24 summarises the finding of the sensitivity analysis and emphasises the impact that gate fees have on B_T. GF-1 has negative B_T for all scenarios and therefore none are viable options as well as having an average variance from initial analysis results of 344%.

GF-2 and GF-4 match the original scenario the closest, with an average variance of 87% and 38% respectively. The higher gate fee in GF-2 also sees LFG-E having the highest ranking for both compliance levels, followed by BSF, VC, AD. The reminder have negative B_T.

GF-3 is the other extreme with all B_T being positive and an average variance of 189%. However, this is the only scenario where status quo landfilling has a positive B_T, further, due to the high gate fees, LFG-E has a significantly higher B_T for both compliant and status quo scenarios. BSF, VC, AD and WC follow in the ranking.

GF-4 and the initial gate fee assumptions produce a similar outcome, but with the B_T of LFG-E being the top ranked in both compliance scenarios due to the gate fees.

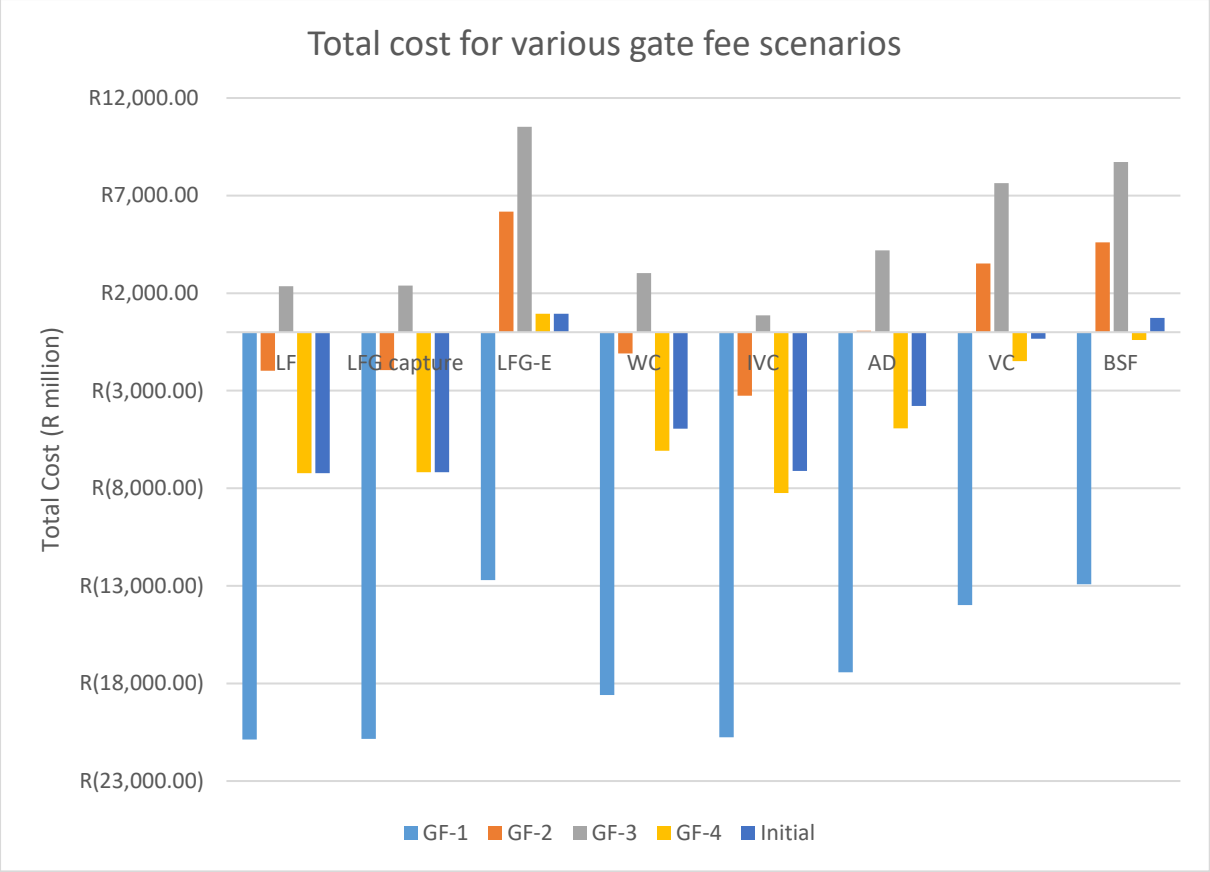

Figure 24: Summarising the impact of various gate fees on the total cost of each scenario.

7.3.3 Combined sensitivities

It is important to check the combined effects of the sensitivity analysis on the final ranking as both these can apply simultaneously. The effects of varying gate fees on the first scenario in LFG-E assumption in the main analysis has been reviewed, the study will now look at the effect on the worst-case assumption, if only LFG-E is viable in large landfills and medium landfills have LFG capture installed (as per national requirements).

Figure 25 summarises the combined sensitivity analysis and it is clear that the ranking of LFG-E has dropped in all gate fee scenarios. In all situations where gate fees for food waste diversion are lower than landfilling, LFG-E is still ranked below BSF and VC. However, LFG-E still ranks higher than AD and WC even in the worst case scenario, which reaffirms its applicability.

The adjusted ranking suggests that LFG-E can provide an increase in total benefit of R3,475 million per year (reduced from R8,167) from traditional landfilling with high compliance levels. BSF, with a range of lower gate fees, can provide an increase to total benefit of landfilling of R6,366 to R7,958 million per year.

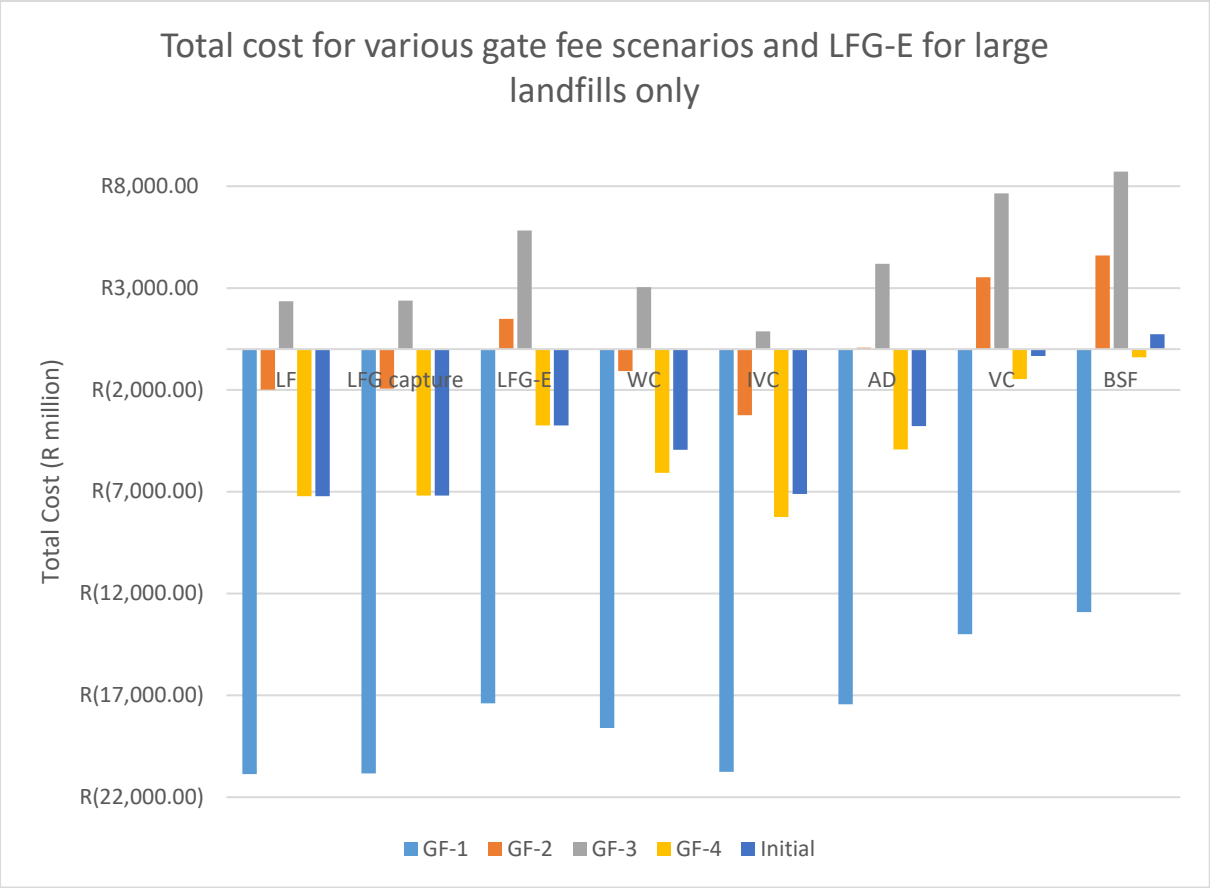

Figure 25: Summarising the effects on total cost of varying gate fees and assuming LFG-E only applies to large landfills.

CHAPTER 8: DISCUSSION

The four pillars of sustainability - economic, ecological, social, and governance - are crucial for a holistic approach to waste management. Behera (2023) emphasises the role of environmental policy, green technologies, and effective waste management in advancing sustainability. Most studies look at optimising either economic or ecological sustainability when analysing potential technologies. However social sustainability and governance needs to be factored in when deciding on applicable options. The importance of community engagement and public participation in waste management practices, as well as ensuring equitable access to waste services, and addressing health and safety concerns related to waste management facilities are crucial for social sustainability (Taelman et al., 2018). Effective waste management requires strong governance frameworks that include clear policies, regulations, and standards for waste reduction, recycling, and disposal. It also involves accountability and transparency in waste management operations and decision-making processes (Jones & Comfort, 2019). Castaldi (2014) underscores the potential of extracting value from waste streams, particularly in developed regions, to enhance sustainability.

8.1 Status quo landfilling in South Africa

The current landfilling practices in South Africa are economically, environmentally, and socially deficient. The majority of landfill sites are uncontrolled or semi-controlled, posing a serious threat to the environment (Remigios, 2010; Idowu et al., 2019). This study estimates that general waste landfills cost the economy, ecology, and surrounding community the equivalent of R8.7 billion per year.

The current state of landfilling in South Africa is characterised by a lack of financial capacity to maintain engineered landfills, leading to their downgrading and conversion to open dumps (Remigios, 2010). This is further exacerbated by the high costs associated with waste management and the structure of containment systems, which are predominantly uncontrolled or semi-controlled (Idowu et al., 2019). Idowu et al. (2019) found that 80% of the documented landfill sites in Sub-Saharan African (SSA) countries were classified as lacking control and containment measures and

that the structure of the containment and controlled regime were identified as important predictors of sustainability.

Despite a long history of recycling in South Africa, the extensive legislative framework and lack of separation at source have made it challenging to drive waste away from landfills (Godfrey & Oelofse, 2017). Godfrey & Oelofse (2017) identify the need for more investment in higher diversion from landfill targets, the presence and impact of the informal waste sector, and the potential opportunities in unlocking social, economic, and environmental benefits from secondary resources. Godfrey and Oelofse (2017) also raise the question of whether South Africa can leapfrog the technology choices of developed countries and move directly towards a stronger materials recovery paradigm.

A range of studies have highlighted the significant health impacts of landfills on surrounding communities in South Africa. Njoku et al. (2019) found that residents living near a landfill in Thohoyandou experienced serious air contamination and health issues, while Moola et al. (2013) identified high levels of hazardous air pollutants, particularly benzene and toluene, at a landfill in Johannesburg. This study attempts to account for these health effects on surrounding populations, in local air pollution external costs, but the link between landfill compliance, size and proximity to populations needs to be further explored.

De Lange and Nahman (2015) estimated the financial cost of landfilling to be R175 per tonne of waste (2022 ZAR), based on gate fees. This is significantly lower than the estimate in this study, which is R484.67, based on the literature. De Lange and Nahman (2015) note that this varied widely between municipalities, as well as these fees not being based on true cost accounting or reflecting the true costs associated with landfill closure. The average weighted gate fee, based on municipal landfills in each province, is estimated to be R376.28 in this study, which indicates that the financial cost assumed by De Lange and Nahman (2015) is an underestimate.

De Lange and Nahman (2015) also estimated an external cost of R183 per tonne of waste (2022 ZAR), based on a landfill in Cape Town, which was then extrapolated across the full volume of food waste. This study, however, estimated a lower external

cost of R135 per tonne of waste. This discrepancy is due to the weighted nature of this assessment, which takes the size, climate region, and proximity to populations of 158 landfills across South Africa into account rather than just Cape Town.

The total estimated cost of disposed food waste was R10.5 billion rand in 2022 ZAR per year, according to De Lange & Nahman's (2015) study. This study estimates the full disposal cost to be R9.2 billion per year. The main discrepancy concerns assumptions on food waste volumes sent to landfills, the potential opportunity cost of the food waste, and the externality cost of landfilling. The revised food waste data by Oelofse et al. (2021), while the same order or magnitude as that used by De Lange & Nahman's (2015) study, assumed different waste volumes per stage in the value chain. This study only looks at the proportion of food waste due to the distribution and consumption phases (3 million tonnes per year), while De Lange & Nahman's (2015) study looked at the entire value chain (10 million tonnes per year). This study assumes food losses during processing get redirected to non-food uses, such as animal feed or oil extraction, enzymes or other components (Oelofse et al., 2021).

This study also includes the compliance levels of each landfill and how they affect the disamenity costs, the GHG benefits of existing LG capture to flaring and LFG-E projects. This study, therefore, recommends the estimated total cost of landfilling to be R619.74 per tonne of waste to be a more accurate reflection of the cost of landfilling in South Africa.

8.2 Compliance and Policy

The compliance of landfills in South Africa has been a topic of concern due to various challenges. These include the co-disposal of different types of waste, lack of proper management, and the impact on waste pickers (Mangizvo & Wiseman, 2012; Schenck et al., 2018). Roberts (2013) suggests that in the Free State, poor waste practices are the norm, and no landfills in the province comply with minimum requirements for landfilling.

It is important to understand the impact of poor waste management practices on the economy, surrounding community and ecology to highlight gaps in governance

further. The disamenity cost encapsulates the effects on human health and nuisance to communities. This study looks at the disamenity cost of landfills on surrounding communities based on South African work by du Preez and Lottering (2009) and Nahman (2011). This is scaled against international literature which estimates the costs for best practice landfill management in urban and rural areas and poor landfill management.

Blight (2006) emphasises how the classification of landfills based on waste type, size, and climate is a key factor in compliance. The study by du Preez and Lottering (2009) only looks at a single landfill in the Eastern Cape, and Nahman (2011) looks at two landfills in the Western Cape; more studies are recommended to understand the effects on different regions in South Africa as well as compliance effects.

The annual municipal landfill audit report done by a South African NGO, AfriForum, in partnership with the DFFE, sheds light on the low compliance levels in each operational general waste landfill across South Africa (De Klerk, 2023). The audit report states that only 17.5% of landfills in South Africa met the required score for compliance. This study uses the audit report to estimate the compliance effects across 158 active general waste landfill sites in South Africa. The disamenity cost to communities surrounding these landfill sites is estimated to be R1.8 billion per year at the status quo compliance levels. Increasing compliance across these sites could save as much as 18% of the total cost of landfilling, or R1.5 billion per year.

The country's waste management sector is in need of significant investment and capacity building to enhance sustainability (Karani & Jewasikiewitz, 2007; Adeleke et al., 2021). The drive towards sustainability in solid waste management in South Africa has led to the promulgation of several legislations and policies directed towards increased efficiency of solid waste management strategies (Adeleke et al., 2021). However, despite the progress in South Africa’s waste management systems over the years, it still faces several challenges and shortcomings, mostly due to poor governance. This makes the transition from a linear economic model to a circular economy, significantly challenging if the basic waste management principles are not achieved (Adeleke et al., 2021).

While the waste hierarchy is embedded in national policy, an extensive legislative framework has made it more and more challenging for the public and private sector to remain compliant and competitive in a local and global market, and still drive waste away from landfill towards reuse, recycling and recovery (Godfrey & Oelofse, 2017). Despite the existence of a legislative framework, the sector faces challenges such as the lack of accurate waste-related data (Godfrey & Oelofse, 2017), this further impacts the potential for waste-to-energy projects, which are hindered by factors such as the lack of sanitary landfill sites, monitoring, funding, and skills (Mbazima et al., 2022). This is emphasised in this study where the viability of LFG-E is considerable with high landfill compliance levels (ranked between 1 and 4) but ranks lower with lower compliance levels.

Godfrey & Oelofse (2017) indicate that the local recycling economy, on par with many developed countries, is in part due to a large and active informal waste sector. In the absence of separation at source across South African cities and towns, informal waste pickers have been key to accessing resources which the private sector has struggled to access, due to gatekeeping by municipalities (Godfrey & Oelofse, 2017). Moving to higher diversion from landfill targets will require more investment by the private sector and by government in the future. The social, economic and environmental benefits of doing this are clear, but must be balanced against the cost that will ultimately be borne by society, as consumers (Godfrey & Oelofse, 2017). The government's waste management policy, while aiming to improve technology, has unintended consequences for landfill waste salvagers (Chvátal & Smit, 2015). Chvátal & Smit (2015) suggest that waste recovery at the landfill remains the best option for meeting waste minimisation goals and the basic needs of the poor, and that the development of an integrated waste management model ensures that destitute waste salvagers regain some form of income security and human dignity.

It must be noted that while the disamenity costs vary with compliance, the GHG emissions, LAP and leachate impacts do not vary. To get a national estimate of the compliance effects on environmental emissions is not possible due to the lack of data on each site. Literature on the effects of leachate from landfills in South Africa indicates a significant impact on water and soil quality, with high levels of heavy

metals and other contaminants posing a threat to both human health and the environment (Ololade et al., 2019; Edokpayi et al., 2018). Rosli et al. (2013) found that closed landfills and poorly managed sites can release significant amounts of methane and carbon dioxide, with the latter posing a health risk. This will vary considerably depending on the type of non-compliance and the climate and regional groundwater and soil profiles. This is an area that needs further research to better understand and quantify the impacts of compliance on emissions.

8.3 Benefits of alternative waste management technologies

A range of studies have explored the potential technologies for food waste diversion from landfills, considering economic, environmental, and social costs. Most LCA studies look at environmental effects, with an emphasis on GHG emissions (Schott et al., 2016). This study found that the effects of GHG emissions had the largest impact of all externalities between technologies.

A review of comparative LCAs of food waste management systems reveals significant variations in global warming potential, largely due to differences in system boundaries, methodological choices, and energy being displaced (Bernstad & la Cour Jansen, 2012; Schott et al., 2016; Yang et al., 2022). These variations are further influenced by the specific food waste management strategy, with the literature generally agreeing that anaerobic digestion is found to be environmentally preferable to composting and landfilling (Yang et al., 2022; Bernstad & la Cour Jansen, 2012; Slorach et al., 2019; Levis & Barlaz, 2011; Gao et al., 2017; Thyberg & Tonjes, 2017). This aligns with this study which ranks the environmental benefit of AD to be the highest of all technologies, followed by VC, BSF, WC and then LFG-E (Chan et al., 2010; Komakech et al., 2015; Perednia et al., 2017). Status quo landfilling has the worst environmental benefit.

Studies that evaluated the environmental and economic potential of food waste technologies abound, but do not take any social factors into account. These studies agree that all forms of composting outperform landfilling (Lu et al., 2020; Bong et al., 2017), which is clear from these results. Further, Slorach et al. (2019) and Fan et al. (2023b) agree that AD outperforms WC in terms of environmental and economic

costs. Slorach et al. (2019) rank LFG-E as the worst solution, even lower than IVC, which is contradicted by Cabaraban et al. (2008) who rank LFG-E over IVC. This discrepancy is due to the different weighting attributed to environmental impacts, weighting given to the LCC, population density around landfills and local conditions in the United Kingdom (Slorach et al., 2019). The LCC analysis by Fan et al. (2023b) agrees with this study in that BSF outperforms AD followed by WC, when combining economic and environmental benefits. Couth et al. (2011), when reviewing the eThekwini Municipality landfill gas to energy project, concludes that small to medium sized landfill gas to electricity CDM projects are not viable in Africa unless there is a renewable energy feed-in-tariff, or unless the gas is simply flared rather than being utilised for power generation.

A few studies found that various forms of food waste composting outperform AD and LFG-E in terms of a combined environmental and economic perspective, even with AD having the lowest environmental impact (Kim et al., 2011; Aye & Widjaya, 2005). These variations can be attributed to the type of analysis, with different weightings given to benefits, different system assumptions and especially low fossil fuel energy mixes in each country. South Africa has both a heavy reliance on coal fired power (hence greater GHG benefits), as well as a shortage of production, therefore a higher preference for clean power producing technologies.

This study is more interested in the evaluation of technologies from a combined economic, ecological, and social perspective. Due to the variance in methodology evaluating MSW solutions, the results can vary considerably and there are several methodologies employed. For comparison with the results in this study, studies that employ environmental cost-benefit analysis (E-CBA), but more specifically a social life-cycle costing (societal LCC) or life-cycle cost-benefit analysis (LC-CBA) (Yang et al., 2022), are reviewed. All studies identified were performed in first world countries with advanced waste collection and treatment systems, which diverges from the South African waste management landscape. However, from the perspective of economic costs (including societal LCC), the ranking of treatment technologies varied a lot from one study to another, attributable to the diverse evaluation methods and different data sources (Yang et al., 2022; Schott et al., 2016).

When comparing the total benefit of traditional technologies like WC, AD and landfilling this study is corroborated in its findings that AD has the highest total benefit (Maalouf & El-Fadel, 2017; Eunomia, 2002). Edwards et al. (2018) revealed that the externality costs of all alternative systems were lower than business as usual (BAU), which in that study is LFG-E and AD. Edwards found that AD has a higher total cost than LFG-E, aligning with this study which finds that LFG-E has a potential higher total benefit than AD (depending on compliance levels, gate fees and base assumptions). This is mostly due to high private costs, and in the case of this study, the higher volume of methane collection due to the larger volume of waste. AD is not considered as viable as BAU or WC in the Edwards et al. (2018) study due to the separate collection required for food waste, high capital costs and single source processing. Edwards et al. (2018) also found that home composting and centralised WC of food waste provide the highest benefits, which diverges from this study, but in Edwards's study, the assumption is that food waste is collected and combined with the existing garden refuse collection and WC systems (Edwards et al., 2018). These separate collection services are not present in South Africa and as such, the variation in existing local waste management services explain the variation in results.

The 2002 Eunomia report prepared for the European Commission did an economic assessment of the options for managing biodegradable municipal waste. With a very similar methodology to this study, its results reinforced the findings of this report, namely that findings vary depending on levels of compliance and best practice; in most countries, LFG-E is preferred over composting (higher benefit), but a transition from status quo landfilling to composting will be beneficial for most countries. The report also suggests that there appear to be external benefits associated with the separate collection of biowastes for composting / anaerobic digestion. The magnitude of these benefits is generally higher for the anaerobic digestion case, where one assumes that the energy generated displaces burdens associated with energy generation from alternative sources (Eunomia, 2002). This is broadly consistent with this study.

Interestingly social LCC studies by Kim et al. (2011) and Martinez-Sanchez et al. (2016) both found that the highest social benefits are for food waste diversion to animal feed, outperforming AD and WC. This is not a technology that was included in

this study but can be comparable to BSF processing which coverts food waste to high value protein assumed to be used in animal feed. In this study BSF processing dominated as having the highest total benefits of all the potential technologies.

No studies could be found that compared the potential of BSF larvae or VC of food waste to other potential technologies in terms of the ecological, social and financial benefits. Lalander et al. (2018) suggest that BSF has a higher potential than AD and WC due to low emissions and high value of products produced, but does not do a full ecological, economic, or social cost accounting. The diversion of food waste from landfills to BSF processing facilities is estimated to provide an additional total benefit of R6.37 to R7.96 billion per year. Depending on the extent of LFG-E installed in South Africa, the other total benefit can range from R3.74 to R8.17 billion per year.

Fan et al. (2023b) note that combined technologies can lead to better cost–benefit efficiency than one treatment technology, aerobic composting or AD, and suggest that combined technologies should be selected for the newly established food waste treatment facilities. The combination of AD and WC or AD and BSF have potential to outperform individual technology choices (Fan et al., 2023b). Lalander et al. (2018) proposes that the inclusion of BSF in existing AD plants for food waste processing have one of the highest opportunity benefits. This is well noted, better results could have been achieved with combined technology scenarios.

Fan et al. (2023b) discussed that combined technologies could lead to better cost–benefit efficiency than one treatment technology. Future analysis should have combinations of technology based on region and food waste volume and combining technologies for value-added by-products. Some examples would be WC and AD, AD and BSF, WC and VC, and BSF and VC.

8.4 Gate fees

The disposal of waste to landfill in South Africa, while being the prominent disposal option, still incurs a cost also known as a gate fee, which is charged per tonne of waste. These gate fees are relatively low in South Africa when compared to more developed economies (GreenCape, 2022b). Despite this, waste generators still

regard landfilling of waste to be a costly exercise. Typically, in well managed landfills, the fees vary depending on the waste being disposed and where it originates, for instance, some landfills might not charge for disposing of inert waste, while waste sourced outside a municipal region might incur higher disposal costs (DEFF, 2020).

Typically, disposal of putrescible waste and general waste (MSW) is charged at different rates, which is due to the additional work required to dispose of putrescible waste and the cost incurred by the landfill to manage methane, odours, and leachate. This rate for the disposal of putrescible waste (expired food waste, abattoir, animal carcasses, used oils, etc.) can be double to ten times higher than the rate for general waste (see Table 11). This study only looks at the rates for general waste disposal, as this is combined with home food and restaurant waste, however it is important to note the potential of diversion of large consignments of food waste and byproducts of the food processing industry that still is diverted to landfill in South Africa.

This safe disposal is of crucial importance for the waste disposer, who is liable for any human health impacts of this waste if not safely treated and disposed. Majority of landfills in South Africa do not have secured premises and communities of waste reclaimers often live in the waste body. Some landfills are entirely run by these reclaimers with no municipal oversite (De Klerk, 2023). These communities play a vital role in the informal recycling economy, but also often live off the incoming waste stream for income and a source of nourishment which has significant potential for damage to human life.

The City of Cape Town (CoCT) has the highest disposal fees, where the full cost of landfill operations was pegged at R640/tonne as per the 2021/2022 fiscal year data, incorporating capital outlays including for post closure, a significant chunk of which was reserved for waste disposal and collection points (CoCT, 2021b). The DFFE (2015) report suggests that the operational expenses of landfills presently oscillate between 150 and 450 ZAR/tonne, contingent on the administrative methods and tariff strategy in place (DFFE, 2015). Generally, waste disposal entrance fees are structured to cover operational expenses, occasionally encompassing capital

depreciation and financial interest charges. Nahman (2011) emphasises that in general there is a lack of understanding among South African municipalities around the financial cost of waste disposal, reflected in the Northern Cape where no gate fees are charged, and no landfills meet minimum compliance requirements.

In fact, as seen in Figure 11, there is a correlation between gate fees and compliance with low gate fees associated with low levels of compliance, the only outlier being the Free State which has relatively high gate fees and low levels of compliance. The impacts and importance of gate fees cannot be understated, as the country's waste management sector is in need of significant investment and capacity building to enhance sustainability, and gate fees will play a crucial role in this (Karani & Jewasikiewitz, 2007; Adeleke et al., 2021).

The average weighted gate fee, as seen in Table 11, is R376.28 per tonne of general MSW, and this was used in this study as the status quo gate fee. Further, in the sensitivity analysis done in 7.3.2, the lowest, break even and highest gate fees were also analysed to fully understand the impact. In the original analysis, gate fees for each technology were assumed to remain the same, assuming separation happened after collection. In the sensitivity analysis gate fees were varied for the diversion technologies with the understanding that these should be lower than landfilling to encourage diversion from landfill.

Gate fees make up 30-45% of the total benefit so the impact on ranking is significant. Higher gate fees promote LFG-E over diversion technologies, but when adjusting for different LFG-E assumptions BSF provides the highest total benefit in all gate fee scenarios. Of note, LFG-E matches or outperforms AD in all scenarios even when adjusted downwards. Accordingly, this study does not recommend AD over LFG-E with respect to food waste diversion: the impact of gate fees plays a significant role in this.

The analysis indicates that the break-even gate fees (R520.59 per tonne), while providing improved financial benefits, still has a negative total benefit due to the external impacts on the environment and society. This is further exacerbated by compliance levels. The highest gate fees (R640 per tonne of waste) see a net

positive for landfilling, which indicates that the externalities are catered for, and post closure costs can be budgeted in the municipality's models. This corroborates Nahman (2011) who indicates that there is a financial lack of understanding among SA municipalities regarding the financial cost of landfilling and a general failure to implement full cost accounting particularly with respect to closure and post closure care, let alone externality costs. Further, no gate fees would see the total cost of landfilling extend beyond R22 billion per year which further exacerbates the sustainability.

The current gate fee structure, as an average across municipalities is not sustainable. This study further reinforces that the loose or complete lack of cost accounting translates into inefficient waste management services as well as hazardous and ecologically toxic landfills. This directly impacts on surrounding communities, local ecology and the people living and working on these sites. The increased gate fees, as seen in the Free State, do not translate to best waste management practices without good governance and any potential benefit can only be achieved with a functional waste management department.

However, while low gate fees indicate a variety of issues, it is important to understand the full impacts of unilateral raising of the gate fees nationally. This approach in itself is not sustainable, even if necessary, as it may encourage illegal dumping which is already a significant challenge in SA. The increased gate fees need to be coordinated with the implementation of effective and affordable waste diversion strategies to encourage the use of these new waste management options as well as reducing the incidences of illegal dumping. Alternatively, incentive-based programmes can encourage the responsible disposal of waste, while gate fees can discourage the easier option of landfilling.

To effectively incorporate full cost accounting principles in municipal waste management, a systematic and unified approach is essential. This approach must enable accurate financial evaluation of various waste management strategies, ensuring comprehensive cost assessments, and incorporate the full costs encompassing the four pillars of sustainability.

CHAPTER 9: CONCLUSION

This research assesses food waste's potential as a resource within South Africa's waste management framework towards sustainable development. By establishing a baseline for waste management costs, it investigates alternative food waste management technologies for their effectiveness in diverting waste from landfills and generating value-added products.

The findings underscore the necessity of a comprehensive strategy that integrates economic, ecological, social, and governance dimensions to transform food waste into energy and other valuable products efficiently. This study illuminates the deficiencies in current landfill practices, underscores the critical role of adherence to regulations and effective governance, and showcases the promising benefits of innovative waste management technologies. It quantifies the annual cost of status quo landfill practices to the economy, environment, and communities at approximately R8.7 billion, advocating for a paradigm shift towards strategic food waste diversion.

In response to the financial limitations faced by municipalities, particularly in Africa and other developing regions, this study highlights the urgent need for adopting best practices in waste management, facilitated by a comprehensive understanding of gate fees and a coherent strategy for full cost accounting, aiming to enhance sustainability. It points out the economic implications of non-compliance and inefficient practices, with a case study from the Western Cape showing the positive impact of adequately priced gate fees on total benefits. A balanced gate fee strategy is recommended to promote waste diversion while addressing existing waste management challenges comprehensively.

The findings advocate for the strategic diversion of food waste as a cost-effective alternative, particularly favouring Black Soldier Fly (BSF) and Vermicomposting (VC) technologies for municipalities, with LFG-E being suitable for larger municipalities in conjunction with additional diversion technologies. In doing so, this approach aligns with South Africa's commitment to Sustainable Development Goal 12.3, aiming to halve food waste by 2030.

Despite the potential for capturing landfill methane and financing sustainable development assets, a gap in holistic program development hampers cost-effectiveness and sustainability. The limited capacity of municipalities to implement effective waste management programs, coupled with financial constraints, calls for significant investment in infrastructure to support sustainable growth (Karani & Jewasikiewitz, 2007).

This study not only revises the cost estimates of landfilling in South Africa but also provides policymakers with a framework to assess waste management options critically. Considering the reliance on assumptions for external cost calculations, these estimates should guide the magnitude of externalities rather than exact figures. The necessity for further research is evident, particularly in generating detailed economic data on the private and external costs of waste management practices and associated damage costs of pollutants.

Furthermore, the impact of disamenity costs — stemming from landfill operations' effects on nearby communities, such as odours, pollution, and decreased property values — cannot be overlooked. The significance of these costs necessitates additional research to explore the implications of landfill compliance, proximity to populations, and the adoption of different waste management technologies on these costs. Understanding and mitigating the disamenity costs are crucial steps towards developing more socially and economically sustainable waste management practices. To catalyse a strategic shift in waste management practices, this paper advocates for detailed regional studies to identify solutions best suited to local conditions. These findings can guide the development of targeted policies that promote efficient, sustainable waste management strategies, enhancing resource recovery, minimising environmental impacts, and improving public health.

In conclusion, this study offers a holistic view of the challenges and opportunities in food waste management within South Africa's waste management sector. It calls for a unified approach to develop and implement sustainable waste management strategies, emphasising the need for further research, investment, and policy development to achieve these goals effectively.

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