ASSESSMENT AND FEASIBILITY OF CONVERTING MUNICIPAL ORGANIC WASTE INTO BIOGAS USING ANAEROBIC DIGESTION: A SOUTH AFRICAN CASE-STUDY

By

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December 2021

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DECLARATION

I, **Gaogane Jephtah Gaogane**, declare that

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Date: December 2021

DEDICATION

Dedicated to:

Loomatsoga, Violet, Michael, Tshegofatso, Loeto, Lesedi, Esther, Kelemogile, Letsweletse, Mpho, Kgalaletso Gaogane

It is of the Lord's mercies that we are not consumed, because his compassions fail not ~ Lamentations 3:22

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ABSTRACT

The present energy crisis in South Africa warrants the need for an alternative and sustainable energy supply. As a sustainable clean energy carrier, biogas has been demonstrated to be a promising renewable energy source for the generation of heat and electricity. The organic fraction of Municipal Solid Waste (MSW) has been reported as a promising feedstock for biogas production and characterisation of MSW is the basis towards successful waste to energy programs. The use of inappropriate equipment and technology choices based on insufficient data on waste volumes and composition have resulted in the failure of many interventions previously introduced in South African municipalities. Assessing the composition and quantity of available biomass for anaerobic digestion (AD), suitable pre-treatment technologies to enhance biogas production as well as optimization of AD parameters such as pH, temperature and substrate ratio were the core components of this research project. The digestate was evaluated and potential use as fertilizer and feedstock for pyrolysis assessed. A mesophilic bench-scale AD of Cow Dung (CD) and Fruit and Vegetable waste (F&V) obtained from a F&V market was conducted. F&V forms the greatest waste in the country, and this facility generates on average 2560 tonnes of waste per annum. This study concludes that utilisation of MSW for AD relies heavily on characterisation data, which is only possible through separation at source. The study recommends the development of a municipal organic waste facility and equally important, diversion of high-end food chains from entering the landfills. Other technologies such as Mechanical Biological Treatment (MBT) can be revisited for possible waste to energy programs at landfill leading to landfill space savings and reduced pollution. Waste pickers at landfills can be employed for this purpose as separation specialists. Bench-scale AD revealed that the benefits of substrate pre-treatment outweighed the effects of co-digestion ratio and pH. Reducing particle size from 1-2mm to <1mm, doubled the methane gas generation in a much shorter time and removed pH induced microbial inhibition in unbuffered reactors. Optimal pH was observed at 7.5-8.5. A co-digestion ratio of 80:20 (CD:F&V) produced higher methane yield for all pH variations in comparison to 60:20. The Digestate measured an average volatile solids loss of 46.4% with a C:N ratio of 12 and a heating value (HV) of 4.30 MJ/kg. Metal analysis of digestate showed presence of Nb, Sr, Si, N, P and K as constituents returned to the soil. The investigation of the digestate as a potential feedstock for pyrolysis yielded a carbon rich biochar with an HV value of 11.5 MJ/kg at 500 ^{o}C and a bio-oil rich in phenols, ketones and carboxylic acids which are important industry products.

TABLE OF CONTENTS

LIST OF FIGURES

LIST OF TABLES

CHAPTER 1- Introduction

1.1 Background and motivation for the research

In South Africa, municipal solid waste management is struggling to evolve into a sustainable system. It has the potential to boost the economy and create employment, thereby promoting sustainable development.

South Africa alone produces over 108 million tonnes of waste per annum (Department of Environmental Affairs, 2018) and 12.7 million tonnes accounts for domestic/household waste (Rodseth, Notten and von Blottnitz, 2020). Of this tonnage, only 10.8 million tonnes (10%) is diverted from landfill (Department of Environmental Affairs, 2018). Consequently, the largest portion is weighed and transported to the landfill without further treatment. Furthermore, the majority of waste in South Africa is not sorted at source, making it difficult to establish the composition of waste disposed (Nkosi, 2014). This presents a major challenge in the handling of waste by municipalities. Most municipalities are committed to addressing this limitation and are continually improving waste management methods. The latest movement is the promotion of waste minimization over waste treatment or landfilling. In the future a countrywide ban on landfilling is imminent and a great deal of waste will, therefore, be diverted from landfill. Municipalities in the Western Cape have impending legislation that state that there will be no landfilling of organic waste by 2027 (Waste: Market Intelligence Report, 2020). In fact, from June 2019 organic waste that has more than 40% moisture content cannot be landfilled. This is because landfill sites are nearing capacity and the capex requirement as well as present legislation does not favour the opening of new landfill sites. This creates a major challenge for municipalities that are already budget constrained. Thus, assessing opportunities to beneficiate MSW is an urgent matter that will have a huge impact for SA from an environmental and economic perspective.

There exist various waste conversion technologies that can be implemented in solid waste management. Selecting the best technology relies largely on the applicability of the process in the South African environment, with emphasis on cost of installation and maintenance. South Africa is a developing nation, with a growing and changing culture and lifestyle. Therefore, the solution should satisfy both current and future trends. Potential waste technologies include recycling, composting, anaerobic digestion, incineration, pyrolysis, and gasification. They form a range of internationally acceptable waste management techniques other than disposal, that ensures minimization, recovery, recycling and/or treatment of waste

(Greben and Oelofse, 2009). This project presents the use of anaerobic digestion as a suitable waste management solution that is available to South Africa. During anaerobic digestion, the organic mass is allowed to ferment in a controlled environment, producing a methane rich gas which is a very efficient fuel source. This process occurs naturally at landfills and is one of the major causes of air pollution and green-house gas emissions from landfill sites. This work therefore seeks to beneficiate this process in order to ensure not only environmental sustainability but also circularity in waste management.

1.2 Problem Statement

Is the organic fraction of municipal solid waste (OFMSW) a suitable feedstock for anaerobic digestion in the South African context?

The present energy crisis in South Africa warrants the need for an alternative and sustainable energy supply. As a sustainable clean energy carrier, biogas has been illustrated as the most promising renewable energy source in the world for the generation of heat and electricity (Demirbas, Taylan and Kaya, 2016). The organic fraction of MSW has been reported as a promising feedstock for biogas production (Igoni et al. 2008, Ayeleru et al. 2016). Characterisation of the MSW is the basis towards successful waste-to-energy programs and it is necessary to understand the following: potential for material recovery, variability in waste composition based on social and temporal factors, facilitation of design processing equipment, determination of the physical, chemical and thermal properties of the waste (Ayeleru et al. 2016, Oelofse et al. 2016). Inadequate information on waste volumes and composition, has resulted in the failure of many interventions previously introduced in South African municipalities as a result of inappropriate equipment and technology choices (Oelofse et al. 2016). Assessing the composition and quantity of available biomass for anaerobic digestion, suitable pre-treatment technologies to enhance biogas production as well as optimisation of the AD parameters i.e., pH, temperature, and feed ratio, will form the core components of the research project. In addition, a detailed characterisation of the digestate produced will be performed and opportunities to use the digestate as fertilizer or as a feedstock for pyrolysis will be examined.

1.3 Aims and Objectives

The aim of this project was to understand the quantity and composition of municipal organic waste that will be used as a feedstock for biogas production in the eThekwini Municipality. In addition, to understand the opportunities and challenges associated with set-up of an anaerobic digestion plant of OFMSW in South Africa and to assess the best possible use of the digestate produced. Understanding the availability of MSW, composition, pre-treatment required and parameters for optimal biogas production will form the main objectives of the study.

The objectives were further broken down into milestones stated below:

Milestone 1: Collection and characterisation of MSW

- Engage with municipalities to obtain existing waste data and acquire data from the South African Waste Information System (SAWIS).
- Plan best strategy for waste collection and characterisation, identify waste categories and sub-categories
- Quantify and characterise municipal solid waste
- Assessment of a mechanical biological treatment (MBT) facility set-up in SA (Challenges and opportunities) for separation of organic and inorganic components.

Milestone 2: Production of biogas using anaerobic digestion

- Using statistically designed experiments, optimise parameters for biogas production.
- Identify co-digestion waste-streams to increase methane yield
- Identify and test pre-treatment options

Milestone 3: Characterisation of digestate and opportunities for beneficiation

- Characterise digestate and establish use as fertilizer or as a feedstock for pyrolysis

1.4 Scope

The experiments in this research were conducted at the Council for Scientific and Industry Research (CSIR) and the University of KwaZulu Natal (UKZN) Environmental Engineering labs, under the SARCHI Chair Waste and Climate Change. Waste characterisation information was obtained from the Clairwood fruit and vegetable market located at 81 Flower road in Durban, KwaZulu Natal, and the Durban Solid Waste (DSW) department of the eThekwini Municipality. Waste material samples were obtained from the Clairwood market and analysed at CSIR and UKZN. The Bio-methane Potential (BMP) of fruits and vegetables samples was conducted in co-digestion with cattle manure. Cattle manure was obtained from a farm near Durban. The following methodological approach was followed in the exploration, identification and execution of the problem statement:

- The literature review was used to establish gaps and understand context for the sturdy
- A representative case sturdy was established. The sturdy was narrowed to Durban, South Africa, which is representative of a large city in South Africa. In Durban, Clairwood market was selected for field studies, and it is the largest Fruit and Vegetable market in Durban.
- A field sturdy was done to conduct waste characterisation of the waste in the area
- A laboratory characterisation was then conducted on the acquired sample
- Finally, a Biomethane Potential test (Anaerobic Digestion) was conducted to established methane potential of the waste

1.5 Outline of Chapters

Chapter 1 – Introduction

- Introduction of the sturdy, project background and rationale
- Aims and objectives
- Scope of the research

Chapter 2 – Literature Review

- Waste management situation in South Africa
- Waste characterisation
- Waste legislation
- Waste disposal technologies
- Anaerobic Digestion technology

Chapter 3 – Case sturdy

- Waste analysis at major landfills in eThekwini municipality
- WROSE model

Chapter 4 – Methodology

- Identification of waste stream
- Characterisation of waste stream
- Lab characterisation

- Methods for conducting BMP
- Methods for conducting pyrolysis of Digestate

Chapter 5 – Results and Analysis

- Characterisation and stream analysis
- Lab characterisation of waste stream
- BMP of organic waste
- Pyrolysis of digestate

Chapter 6

- Conclusion and recommendations

Chapter 7

- References

Chapter 8

- Appendices

1.6 Chapter summary

This introductory chapter provided a description of the problem statement, briefly explaining the relevant background and motivation for the study and furthermore stating the aims and objectives. The next chapters will provide insight into the Anaerobic Digestion technology as a proficient solution to waste management as well as demonstrate its use in the generation of methane, a combustible gas with the potential to use as a fuel alternative to fossils. The next chapter provides the literature review of the study, which will provide insight on the study undertaken through analysis of existing information on the subject. It is through the literature review that a gap was identified and explored.

CHAPTER 2 - Literature Review

2.1 Introduction

This chapter covers the literature available on anaerobic digestion as one of the solutions in managing solid waste in a sustainable manner. The various sections cover important topics that bring to light the current waste management practices in South Africa which is representative of a typical developing nation with great potential in management and especially in the circularisation of waste.

Section 2.2 discusses in a broad sense the municipal solid waste management in South Africa. This section presents the shift in waste treatment strategies and concepts to globally accepted practises that minimises waste generation rather than put emphasis on waste disposal. The section discusses waste generation factors as well as compositional analysis of the South African waste stream. The data is compared to the rest of the world. The assessment of waste disposal trends as well as the legislation available to the country is conducted. Lastly, the section introduces the waste disposal technologies.

Section 2.3 introduces anaerobic digestion (AD) as a solution to the waste management situation in South Africa as a developing country. The section covers the history of anaerobic digestion (AD) and relevance of the technology in the present age. The section further looks at the basics of AD and furthermore explains how the process of AD works. The parameters affecting AD performance are stated and discussed. The organic feedstock available to South Africa as well as the biomethane potential of the individual substrates is presented and discussed. Lastly the types of reactors and reactor configurations in AD are assessed. These are as critical to the success of AD process as well as the parameters affecting the microbial activity of the process.

2.2 Municipal solid waste management

Solid waste is a low liquid material that arises as a by-product of human activities and is bound for disposal. Waste generation occurs when an item perceived to have lost value is stored or collected (Statistics South Africa, 2018). It is the responsibility of the municipality to ensure that waste is collected, transported, monitored and disposed of in a manner that is in compliance with the rules and regulations of the country. This forms the basis of a municipal solid waste management system, which implements the necessary procedures, precautions and actions that should be followed during the handling of waste from collection to disposal.

A typical management system would involve practices such as collection, waste characterisation and disposal as well as promotion of waste minimization, re-use, and recycling of waste. Waste characterisation is an important aspect of waste management, as it provides a means for understanding solid waste generation trends in various demographics and assists in developing a better waste management system and infrastructure for improved handling of waste (Western Cape Government, 2017).

2.2.1 Municipal waste management in South Africa

Waste Management is defined by Statistics South Africa (2018), as the “collection, transportation, processing or disposal, managing and monitoring of waste materials”. The Department of Environmental Affairs (2012) separates waste into two classes, namely general or hazardous waste. General waste does not pose an immediate threat to health or the environment. Examples include domestic waste, business waste, waste from building and demolishing as well as inert waste (Statistics South Africa, 2018). Hazardous waste on the other hand is generally chemical in nature and presents a potential risk to health or the environment. The South African National Environment Management: Waste Act (RSA, 2009), defines hazardous waste as follows: “any 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”. The definition provided above relates to waste in any form, either solid, liquid, gaseous or radioactive. The management of solid waste is a vital move towards sustainable development.

South Africa is committed to sustainable development. For the waste sector, this involves monitoring raw material usage, resource efficiency and product design, waste prevention or waste minimization where applicable. Figure 2.2-11 below depicts the municipal solid waste management system. The flow of waste material from generation is structured to minimize the amount of waste that may end up discarded in the landfill. It incorporates the minimization strategy of reducing, reusing and recycling of waste. This approach is motivated by the hierarchical approach to sustainability shown in Figure 2.2-2, adopted by the National Environment Management Act (Act 59 of 2008). The figure illustrates the preference for waste prevention and minimization (reduce, reuse and recycle) and moves away from processes such as waste treatment and landfilling.

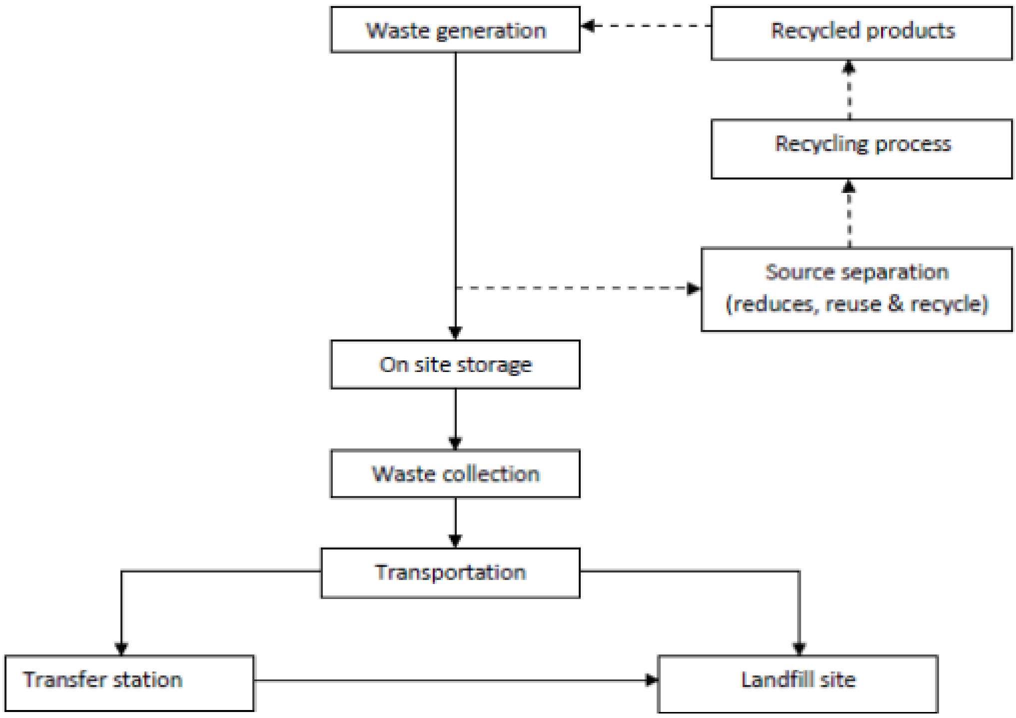
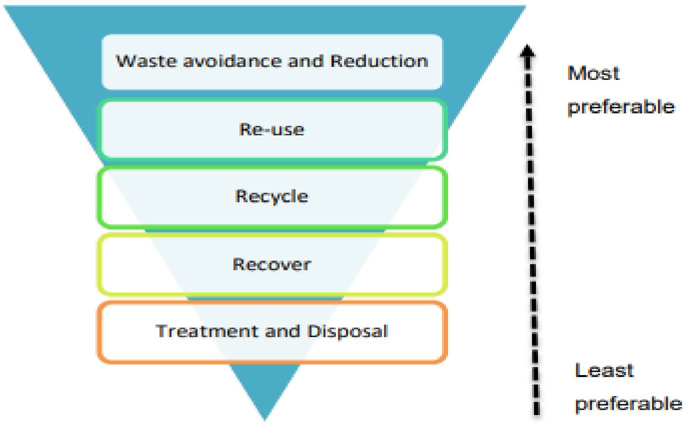


Figure 2.2 - 1: Municipal solid waste management system (Department of Environmental Affairs, 2012)



Source: Adapted from DEA (2011, 2012)

Figure 2.2 - 2: Waste Hierarchy (Department of Environmental Affairs, 2012)

Waste generation is a function of several factors and varies significantly for different communities (urban or rural) and level of awareness (education). This is especially true when monitoring the amount of waste generated per province in South Africa, compared to the provincial population. Gauteng province is the most populated region in South Africa, consequently it generates the greatest proportion of waste (45%) (Table 2.2-1 and 2.2-2) (Oelofse, 2015) . This creates a clear link between population growth and waste generation, as with an increase in population, the demand for and consumption of resources increases and consequently, the generation of waste. A steady increase in the total population of South Africa has been observed over the past 10 years (Table 2.2-2). This growth may be due to factors such as rural-urban migration caused by the need for employment and resources, particularly amongst the youth, and by the cross border migration motivated by the collapse of economies due to political instability and civil wars (Naidoo, 2009).

Other potential drivers of waste generation in South Africa include economy, income per household, urbanisation and globalisation (Department of Environmental Affairs, 2018).

Table 2.2 - 1: Proportion of municipal waste by province in South Africa, 2011 (Department of Environmental Affairs, 2012)

Province	kg/capita/annum	Waste generated as % of Total waste
Western Cape	675	20
Eastern Cape	113	4
Northern Cape	547	3
Free State	199	3
KwaZulu Natal	158	9
North West	68	1
Gauteng	761	45
Mpumalanga	518	10
Limpopo	103	3

Table 2.2 - 2: Population per province (Department of Environmental Affairs, 2012)

Province	Total population (thousand)									
	2002	2003	2004	2005	2006	2007	2008	2009	2010	2011*
Western Cape (WC)	4646	4755	4859	4964	5071	5162	5258	5369	5468	5823
Eastern Cape (EC)	6521	6541	6558	6574	6587	6612	6633	6649	6656	6562
Northern Cape (NC)	1088	1098	1106	1115	1123	1131	1140	1148	1154	1146
Free State (FS)	2777	2795	2811	2826	2842	2863	2884	2905	2919	2746
KwaZulu Natal (KZN)	9683	9802	9915	10024	10134	10242	10348	10461	10551	10267
North West (NW)	3227	3261	3294	3325	3357	3389	3421	3454	3479	3510
Gauteng (GT)	9189	9387	9577	9766	9961	10142	10333	10556	10754	12272
Mpumalanga (MP)	3391	3430	3464	3493	3519	3546	3576	3610	3639	4040
Limpopo (LP)	5011	5048	5081	5111	5138	5171	5201	5230	5250	5405
Total	45533	46117	46665	47198	47732	48258	48794	49382	49870	51771

Economy

A growing economy presents new enriching opportunities, and finances and resources are channelled appropriately to cover the needs of the society. An increase in resource usage results in an increase in waste generation, either directly through manufacturing industries or indirectly through higher incomes (Department of Environmental Affairs, 2018). Conversely, a failing or collapsed economy results in less finances and spending and therefore less monetary support for municipal solid waste management. Recessions and sudden disease pandemics such as Covid19 in 2020 could force the country to divert finances and furthermore instigate a shut down in order to curb the effects of the outbreak, which would significantly affect waste management financing and management.

Typically, the economic performance of a country is determined using the Gross Domestic Product (GDP) or the monetary value of all goods and services produced in a given period (Department of Environmental Affairs, 2018). South Africa recorded a growth in GDP from the year 2013 to 2017. The South African GDP was over R3.5 billion in 2013 and increased to over R4.6 billion in 2017 (Figure 2.2-3). Provincially, Gauteng contributed the highest percentage GDP with 35% in 2016 followed by KwaZulu-Natal and Western Cape with 16% and 14% respectively (Figure 2.2-4). Since not all waste is common, and may be specific to certain sectors, it is important to understand the contribution of such sectors to the GDP. Major wate generators such as manufacturing and construction as well as mining and quarrying accounted for a total of 25% of the GDP. (Figure 2.2-5). The highest producer of organic waste, which is a department of agriculture, forestry and fishing had a constitution of 8% to the GDP.

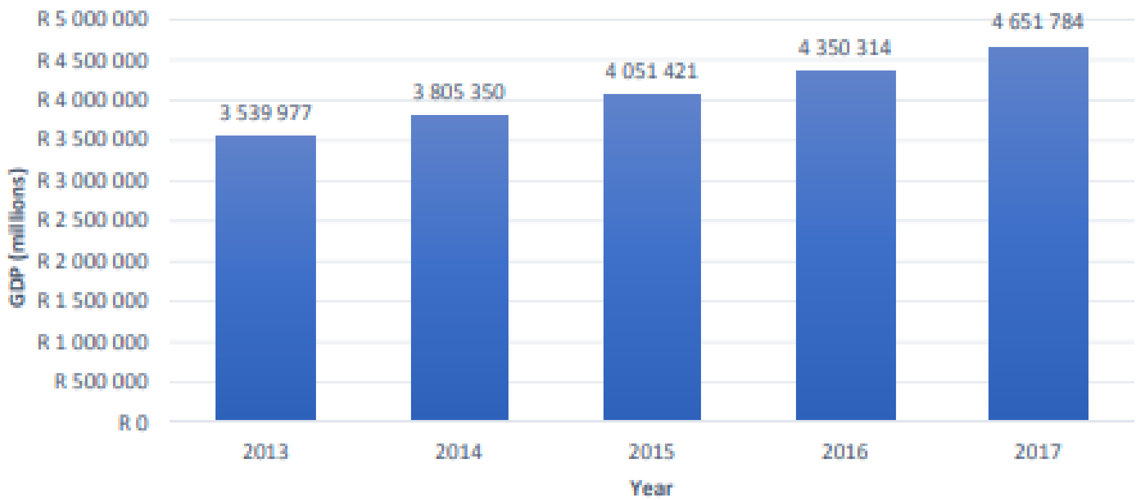


Figure 2.2 - 3: South African GDP from 2013 to 2017 (Department of Environmental Affairs, 2018)

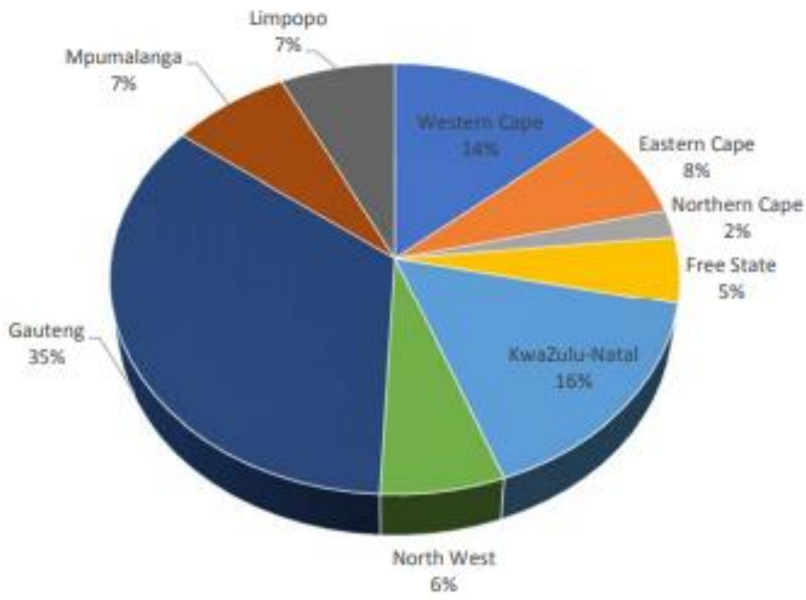


Figure 2.2 - 4: Provincial contribution to national GDP in 2016 (Department of Environmental Affairs, 2018)

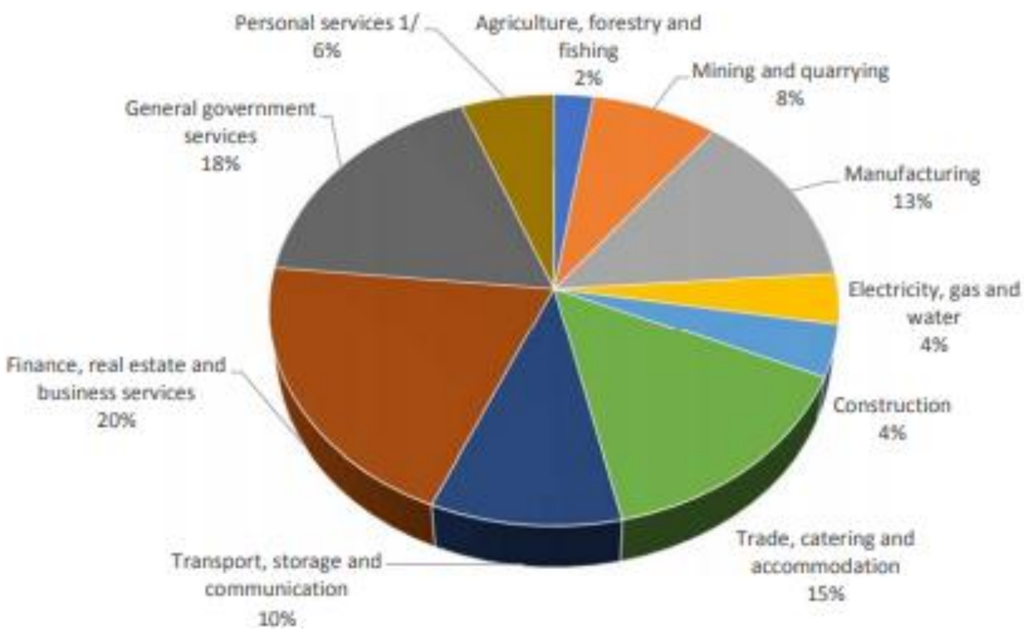


Figure 2.2 - 5: GDP contribution of key economic sectors in 2016 (Department of Environmental Affairs, 2018)

Income per household

The level of income has a major influence on the standard of living of individuals or households, which subsequently influences the consumption of goods and services and therefore the amount of waste generated. Of the 20 million tonnes of municipal solid waste generated in 2011, the waste generation per capital was calculated to be 0.41, 0.74, 1.29 kg/person/day for low, medium and high-income groups, respectively (Oelofse, 2015) (Figure 2.2-6). This shows that low-income groups are the least generators of waste. However, since the low-income groups reside in rural communities where waste management services are poor in a developing country, waste pollution is mostly associated with this group. The vast majority of households in South Africa fall under the low-income bracket, with the Western Cape having the highest percentage (95%) of low-income households in 2011.

Urbanisation

Urbanisation and economic development are closely associated. Municipal Waste Management (MSW) is generally known to be an urban issue, as most waste is generated in urban areas (Department of Environmental Affairs, 2018). The need for employment and better opportunities prompts youth to migrate from rural to urban areas. It has been reported that two thirds of South African youths live in urban areas, and by 2030 the proportion of people living in urban areas would be 71.3%.

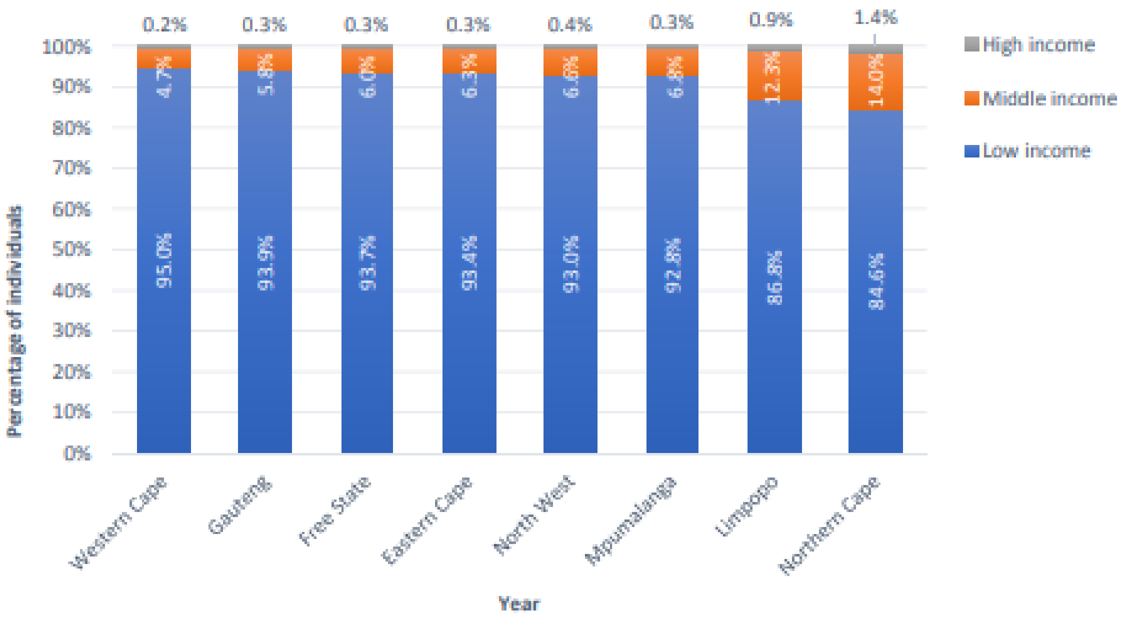


Figure 2.2 - 6: Distribution of individuals from low, middle and high income areas in 2011 (Department of Environmental Affairs, 2018)

2.2.2 Municipal waste composition in South Africa

In 2011, South Africa generated over 108 million tonnes of waste per annum, of which 59 million tonnes was general waste (55%), 1 million tonnes was hazardous waste (1%) and 49 million tonnes was unclassified waste (44%) (Figure 2.2-7) (Oelofse, 2015; Statistics South Africa, 2018). Unclassified waste, in this case, refers to waste that may qualify as both general and hazardous. In 2017, 42 million tonnes of general waste was generated, and hazardous waste accounted for 38 million tonnes, followed by 27.8 million tonnes of unclassified waste (Department of Environmental Affairs, 2018). The recycling rate of general waste was 10%, and hazardous waste was calculated at 7% (Department of Environmental Affairs, 2018). A breakdown of general waste for 2017, showed 'Other' as the largest portion of waste generated (35 %) (Figure 2.2-8). 'Other' general waste comprises biomass from sugar mills, sawmills and the paper and pulp industry (Department of Environmental Affairs, 2018). 'Other' waste is followed by organic waste with 16% and construction and demolition waste with 13%. A more detailed breakdown of the composition of general waste stream is shown in Table 2.2-3. The table shows that in 2017, 11% of general waste was recycled compared to 10% recorded in 2011. In addition, 12% of organic waste was recycled, and over 5.8 million tonnes was sent to landfill. The highest recycling rate in 2017 was recorded for the metal industry at 48%. All municipal, commercial, and industrial waste and other waste generated was sent to landfill.

Figure 2.2 - 7: Waste Composition in RSA, 2011 (Department of Environmental Affairs, 2012)

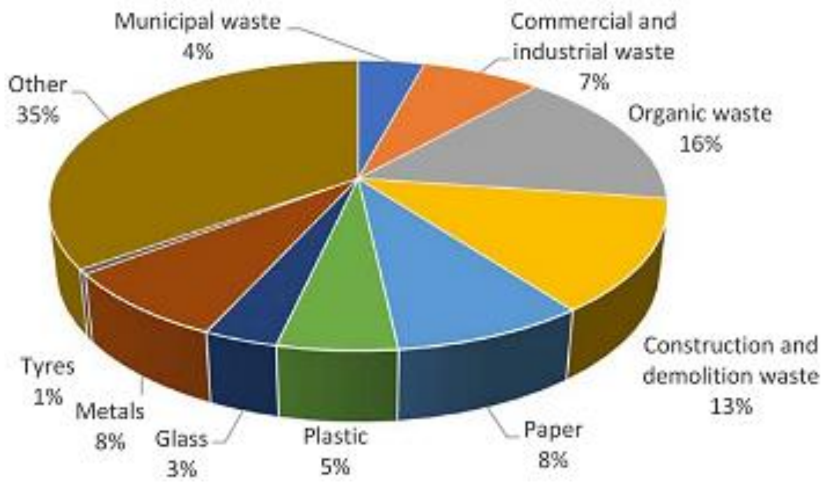


Figure 2.2 - 8: General waste breakdown for 2017 (Department of Environmental Affairs, 2018)

Table 2.2 - 3: Waste management for various sources in 2017 (Department of Environmental Affairs, 2018)

Waste type		Generated	Recovered / Recycled	Landfilled	Percentage Recovered / Recycled
GW01	Municipal waste	1 770 009	-	1 770 009	0%
GW10	Commercial and industrial waste	3 179 157	-	3 179 157	0%
GW13	Brine				
GW14	Fly ash and dust				
GW15	Bottom ash				
GW16	Slag				
GW 17	Mineral waste				
GW 18	WEEE				
GW 20	Organic waste	6 656 234	812 206	5 844 028	12%
GW 21	Sewage sludge				
GW30	Construction and demolition waste	5 360 556	305 761	5 054 795	6%
GW50	Paper	3 635 825	1 414 378	2 221 447	39%
GW51	Plastic	2 247 323	332 713	1 914 610	15%
GW52	Glass	1 395 103	320 000	1 075 103	23%
GW53	Metals	3 345 565	1 622 059	1 723 506	48%
GW54	Tyres	221 751	64 061	157 690	29%
GW99	Other	14 868 997	-	14 868 997	0%
	Total general waste (t)	**42 680 520**	**4 871 178**	**37 809 341**	**11%**

A minimal number of studies have been undertaken on waste characterisation in different municipalities (Department of Environmental Affairs, 2012). Green and garden waste were combined to represent organics generated (Figure 2.2-9) (Department of Environmental Affairs, 2012). The data obtained for Cape Town and Gauteng showed similar proportions of the different waste reported. Organics were 15% and 18% for Gauteng and Cape Town, respectively. The largest percentage of waste was the non-recyclables which accounted for 38% and 40% for Cape Town and Gauteng, respectively. Waste characterisation by source information was obtained for the year 2004 for 4 municipalities (Figure 2.2-10). No data was found for KwaZulu Natal municipality. In the case of Johannesburg, it was presumed that waste categorized as 'other' and illegal dumping originated from household sources and was therefore classified as household waste (Department of Environmental Affairs, 2012). Of the 4 municipalities investigated in Figure 2.2-10, only Johannesburg recorded commercial industries as the highest source of waste, whilst the other 3 municipalities reported households as the main source of waste.

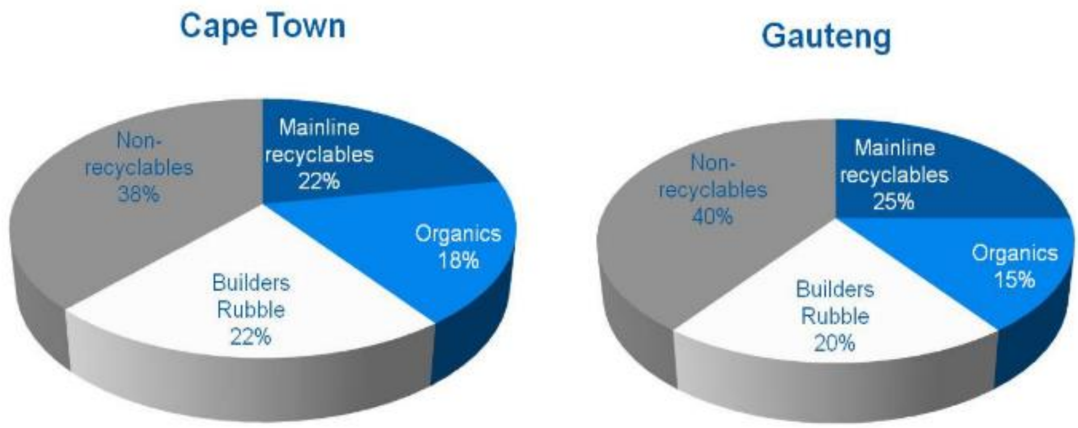


Figure 2.2 - 9: Composition of Municipal waste (by mass percentage) (Department of Environmental Affairs, 2012)

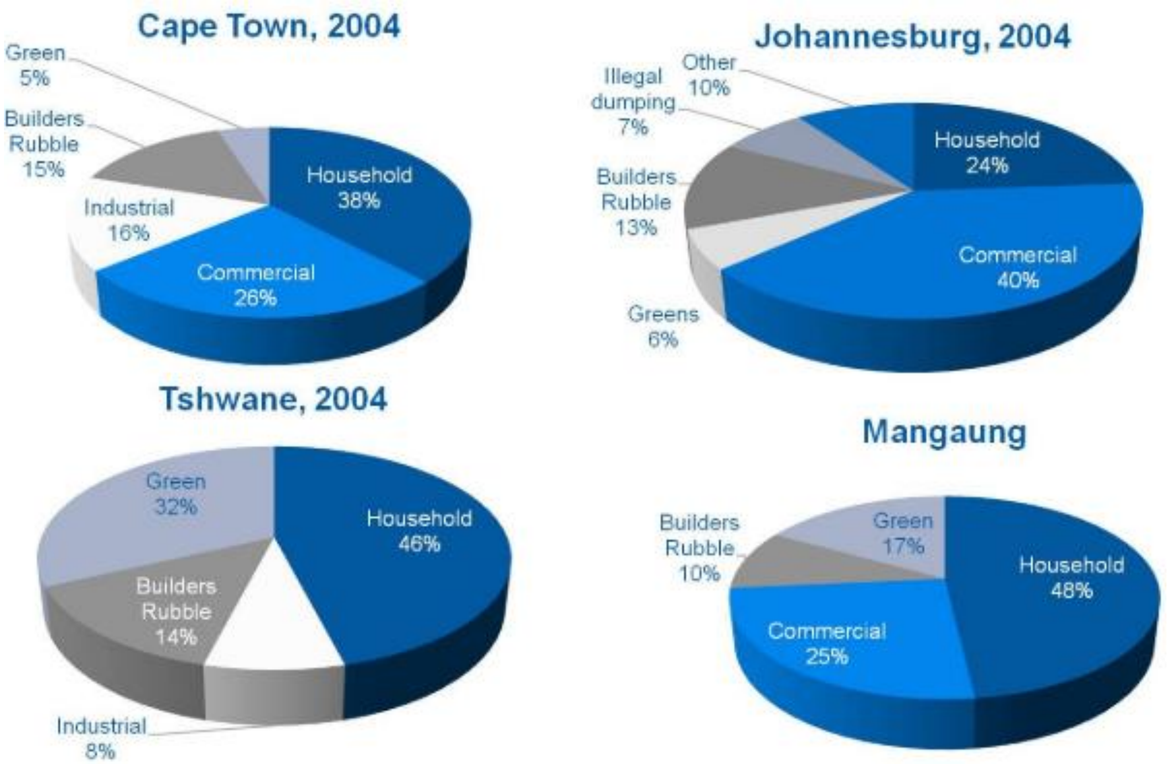


Figure 2.2 - 10: Composition of Municipal waste by source (Department of Environmental Affairs, 2012)

Table 2.2-4 provides a more detailed account of the domestic waste composition as weight percentages generated by different municipals. Organic waste was recorded as the highest generated group in most municipals. The composition of organic waste ranged from 10% in Northern Cape to 58% in the North-West municipality.

Table 2.2 - 4: Waste composition for various municipals (mass percentage) (Oelofse, 2015)

	Paper	Plastic	Glass	Metal	Organic	Other
Cape Town	24	15	12	7	32	9
Western Cape	24	22	9	8	22	16
Northern Cape	21	18	10	3	10	39
Free state	14	12	8	6	42	19
North West	11	9	6	2	58	14
Johannesburg	17	10	5	3	36	30

2.2.3 Waste disposal trends in South Africa

Around 80% of all waste goes to landfill in South Africa (Department of Environmental Affairs, 2018). About 11% of legal waste disposal facilities are treatment facilities, and the majority of these are wastewater treatment works (WWTW) and facilities for treatment of healthcare risk waste (Department of Environmental Affairs, 2018). South African law identifies landfill sites as one of the solutions for waste disposal, and several areas have been legally designated. Generally, South African landfills accept domestic, commercial and non-hazardous industrial, building and garden waste (Tawona, 2015). The main problem surrounding landfilling is the emergence of illegal landfill sites which not only inhibit the operation of a structured management system but is also a health hazard to the community. Legal landfill sites are strategically located in well-established facilities and placed at a safe distance from the public. Furthermore, facilities contain defined boundaries and limitations that are adhered to for safe operation.

South Africa has just over 43% of legally registered landfills in the year 2011. Table 2.2-5. There percentage of registered general waste landfills and recycling facilities are 44% and 22%. The South Africa state of the waste report reported the number of landfill sites indicated in Figure 2.2-11 for the year 2018. Most landfills are located in highly populated provinces such as Gauteng and KwaZulu-Natal, as populated areas tend to generate more waste. The legal landfill sites merely represent a fraction of the number of landfills that actually exist. Table 2.2-6 shows that General waste landfill sites represent less than 44% of the actual number of landfills. Of the 1336 recorded waste disposal facilities, only 583 are legally permitted (Table 2.2-6). Of the landfills represented in Figure 2.2-11, 473 of these landfills are accessible to the public (dump sites), 269 are inaccessible to the public (on-site dump sites), and 49 are onsite storage facilities (Department of Environmental Affairs, 2018).

Owing to the reliance on landfilling, the remaining capacity of landfill sites remains a great challenge for some municipalities. The estimated remaining capacity for the cities such as Cape Town and Johannesburg are 5 and 8 years, respectively (Figure 2.2-12). This raises the need to divert waste from landfill.

Despite there being many landfill sites and waste deposition is seemingly the most utilised method in the country, it is still the least preferred in the waste management hierarchy. There exist several waste treatment and recycling facilities in the country, which may possibly take over the majority of waste handling in the future. Figure 2.2-13 shows the number of licenced waste recycling and recovery facilities found in different provinces in the country, most of which are privately owned (Department of Environmental Affairs, 2018). Gauteng has the largest number with 30 facilities followed by Western Cape with nine. Thirty seven of these licenced facilities sort, shred, grind and bail waste and including scrap metal, 11 of them scrap and recover motor vehicles, and 18 recover specific wastes such as oils, solvents and tyres (Department of Environmental Affairs, 2018).

The number of waste treatment facilities per province in South Africa is shown in Figure 2.2-13. Seventy one of these are sewage and wastewater treatment facilities, 26 are for treatment of healthcare risk waste (HCRW), 11 for the physical or physio-chemical treatment of waste, four are composting facilities and only one licensed for biogas installation (Department of Environmental Affairs, 2018).

Table 2.2 - 5: Number of legal landfill sites by province (Muzenda, Ntuli and Pilusa, 2012)

Provinces	Legal landfills
Western Cape	97
Eastern Cape	120
Northern Cape	103
Free State	67
KwaZulu-Natal	119
North West	35
Gauteng	160
Mpumalanga	72
Limpopo	44
Total	817

Table 2.2 - 6: Various waste management facilities in RSA (Muzenda, Ntuli and Pilusa, 2012)

Type of Landfills	Number of facilities	Number of Permitted facilities	Percentage of legal facilities
General Waste Landfill sites	1203	524	43.56
Hazardous Waste Landfill sites	77	41	53.25
Medical Waste storage facilities	12	4	33.33
Recycling facilities	9	2	22.22
Transfer stations	35	12	34.29
Total	1336	583	+

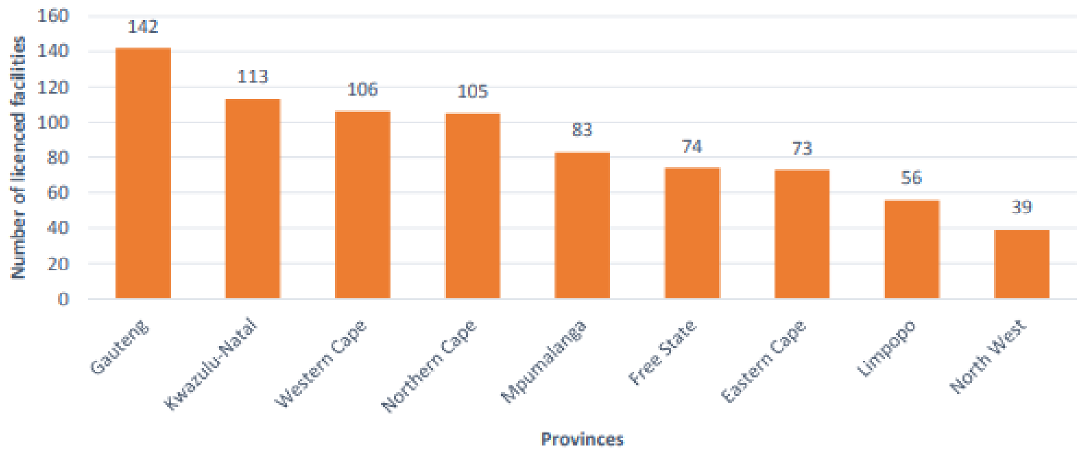


Figure 2.2 - 11: Waste treatment facilities for various provinces in RSA (Department of Environmental Affairs, 2018)

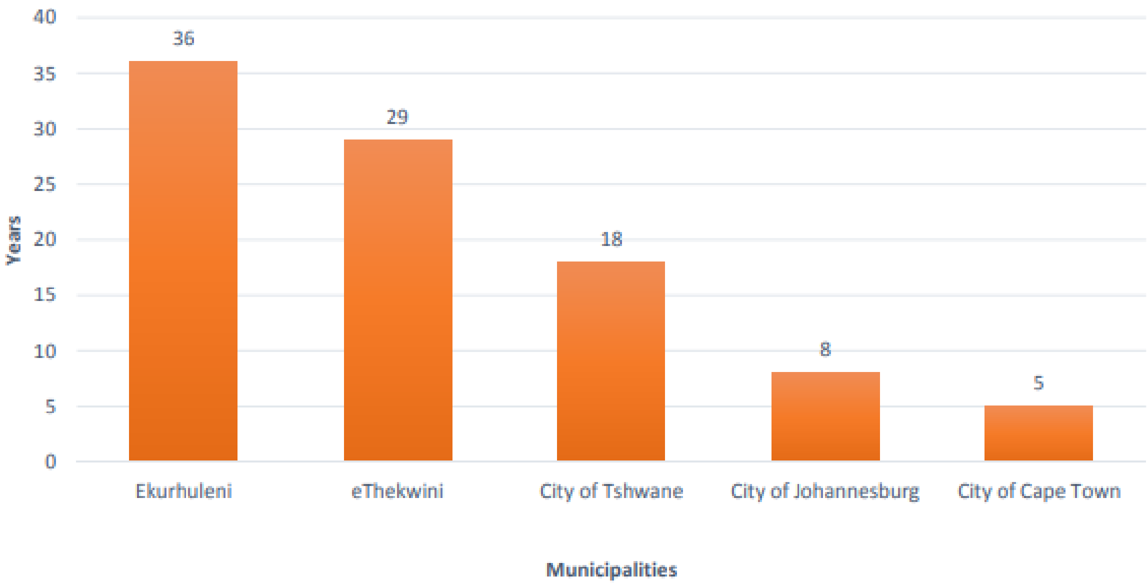


Figure 2.2 - 12: Estimated remaining landfill space for South Africa's largest municipalities (Department of Environmental Affairs, 2018)

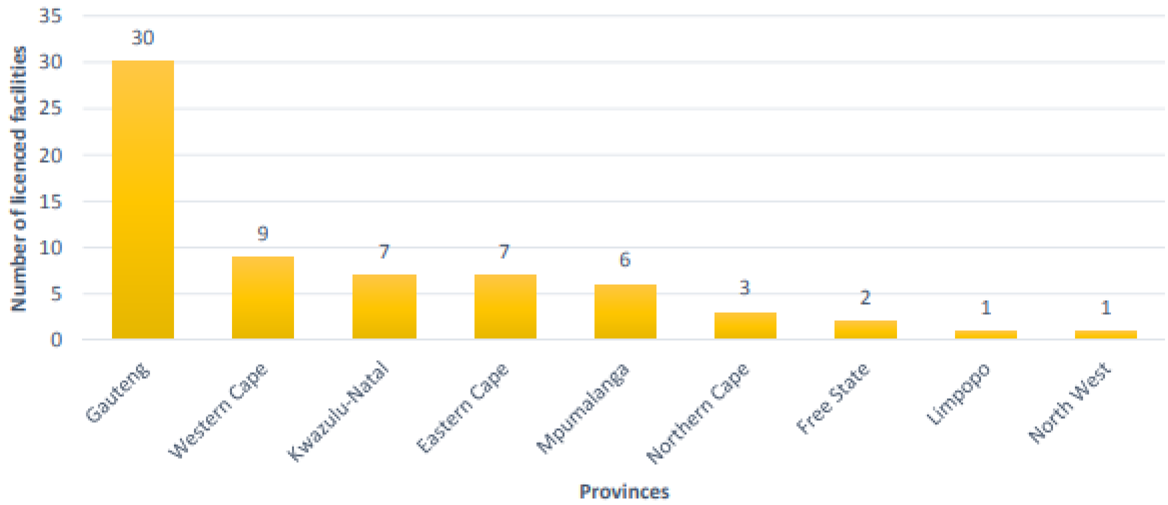


Figure 2.2 - 13: Number of waste recycling and recovery facilities per province in South Africa (Department of Environmental Affairs, 2018)

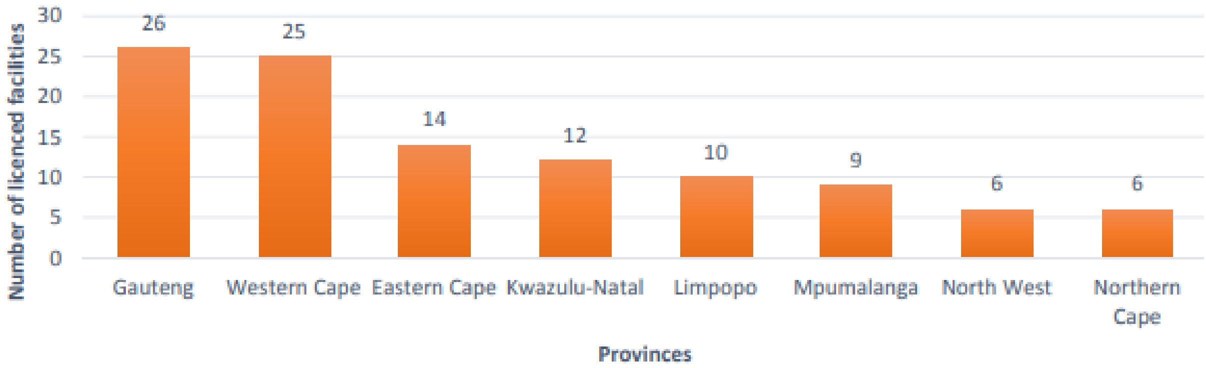


Figure 2.2 - 14: Number of waste treatment facilities per province (Department of Environmental Affairs, 2018)

2.2.4 The legal framework on waste management in South Africa

The legislation for waste management was established in South Africa in 1989, with emphasis on waste management, and human and environmental safety (Figure 2.2-15). The first and most basic legal framework governing the disposal of waste in South Africa was the Environmental Conservation Act (Act No. 73 of 1989) (ECA) (Department of Environmental Affairs, 2018). This Act “required any type of waste facility including transfer stations, storage facilities and recycling plants etc., to be regarded as a deposit site of which a ‘Section a 20(1)’ ECA permit was required” (Department of Environmental Affairs, 2018). The ECA Act (Act No. 73 of 1989) was amended a few times and later revoked in its entirety as it was said to have been “unsuccessful and inadequate” (Department of Environmental Affairs, 2018).

This brought about the introduction of the National Environmental Management Act (NEMA) of 1998 (Act No. 107 of 1998) (Muzenda, 2014). . The most significant changes introduced were the redefining and addition of important terms as well as provision for temporary waste storage (Muzenda, 2014). It allowed for the administration of environment management legislation in a progressive manner (Department of Environmental Affairs, 2018). NEMA (Act No. 107 of 1998) ensured that permits for waste disposal sites were issued, a manifest system for the transportation of hazardous waste was developed, and hazardous waste generators and transporters were registered (Muzenda, 2014). The underlying fundamental principles of NEMA include, “polluter pays, cradle to grave, the precautionary principle and waste avoidance and minimization” (Department of Environmental Affairs, 2018). Under the NEMA Act, a legal framework was developed around non-landfill waste technologies (Department of Environmental Affairs, 2018). Before a permit was issued, the minimum requirements needed to be met. These were the minimum procedures, action and information required to obtain the permit (Muzenda, 2014).

Figure 2.2 - 15: Legislative timeline leading to State of Waste Report (SoWR) (Department of Environmental Affairs, 2018)

The National Environmental Management: Waste Act 2008 (Act No. 59 of 2008) (NEM: WA) came into effect in the 1st of July 2009 (Department of Environmental Affairs, 2018). This Act is waste specific and is considered the most pro-active in regulating waste. The NEM: WA promotes the aforementioned principles of waste hierarchy which are internationally renowned practices that have since been adopted by the country (Department of Environmental Affairs, 2018). The waste hierarchy evolved since the 1999 National Waste Management Strategy (Figure 2.2-16), placing greater emphasis on waste avoidance and minimization in the 2010 strategy. The 2010 strategy includes the National Environmental Management: Waste Amendment Act (Act No. 26 of 2014) (NEM: WAA) which came into effect on the 2nd of June 2014 to fulfil the shortcomings of the NEM: WA (Department of Environmental Affairs, 2018).

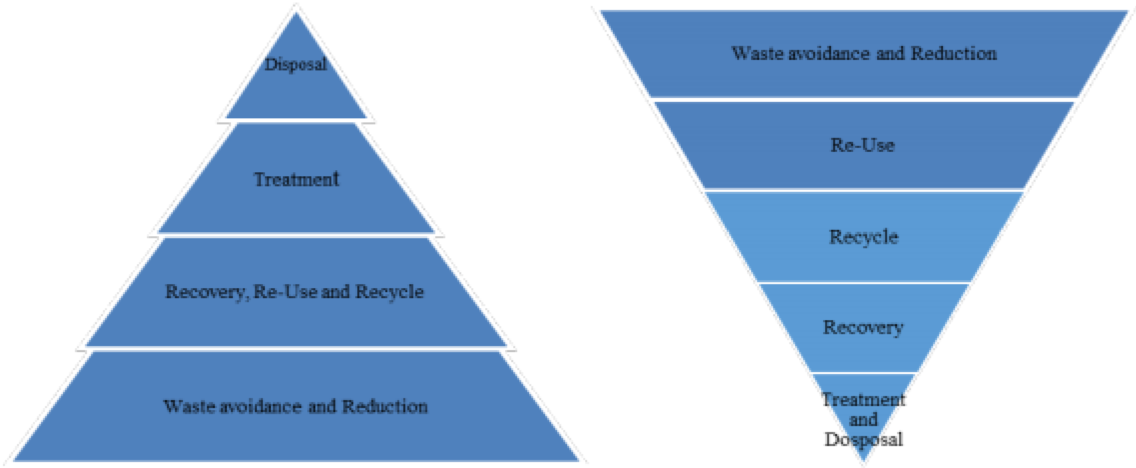


Figure 2.2 - 16: Waste Hierarchy in 1999 (left) and in 2010 (right) (Department of Environmental Affairs, 2018)

2.2.5 Municipal waste management in the rest of the world

"The world generates on average 0.74 kg of waste per capita per day, but the national waste generation rates fluctuate widely from 0.11 to 4.54 kg per capita per day" (Kaza *et al.*, 2018). The global generation of waste is 2.01 billion tonnes per annum, and at least 33% of that waste is not managed in an environmentally safe manner (The World Bank, 2020). High-income countries, though only accounting for about 16% of the world's population, generates over 683 million tonnes of waste (The World Bank, 2020). This is at least 34% of the global waste generated per annum. Global waste generation is expected to grow to 3.40 billion tonnes of waste by 2050, with levels from low-income countries tripling in volume (Kaza *et al.*, 2018). Waste generation for high-income countries is expected to grow by 19% whilst low, and middle-income countries are expected to grow by 40 % (The World Bank, 2020). The share of waste generated per region by percentage is shown in Figure 2.2-17 and by tonnage in Figure 2.2-18, with projected waste generation demonstrated in Figure 2.2-19. The smallest generators of waste are the Middle East and North Africa (6%), and the largest generators are East Asia and the Pacific region (23%). Figure 2.2-20 illustrates waste generation per income level. Low-income class generates the least amount of waste at 5% of the total waste. The upper middle class and high class generates a combined 66% of waste which accounts for 1.241 billion tonnes of waste per annum.

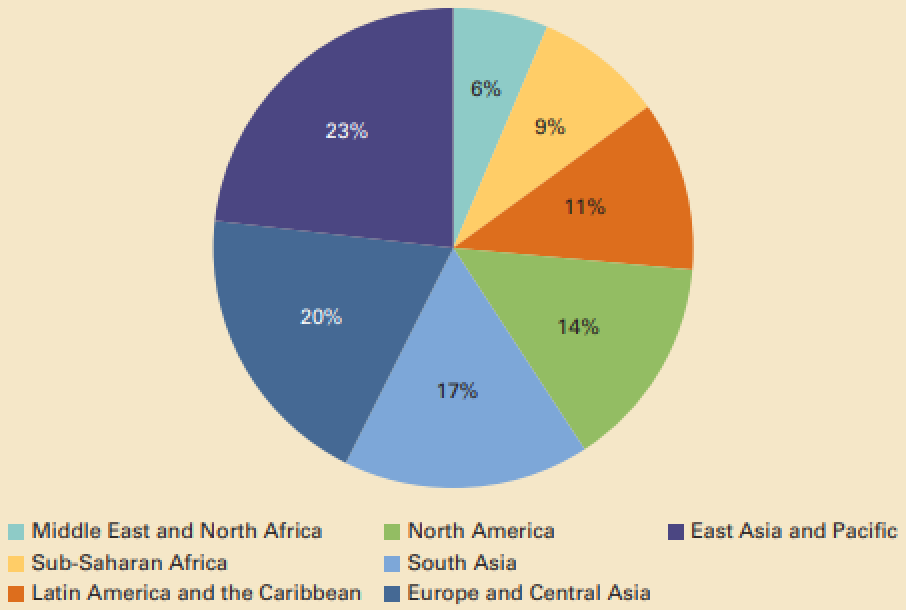


Figure 2.2 - 17: Waste generation by Region (percentage) (Kaza *et al.*, 2018; The world bank, 2020)

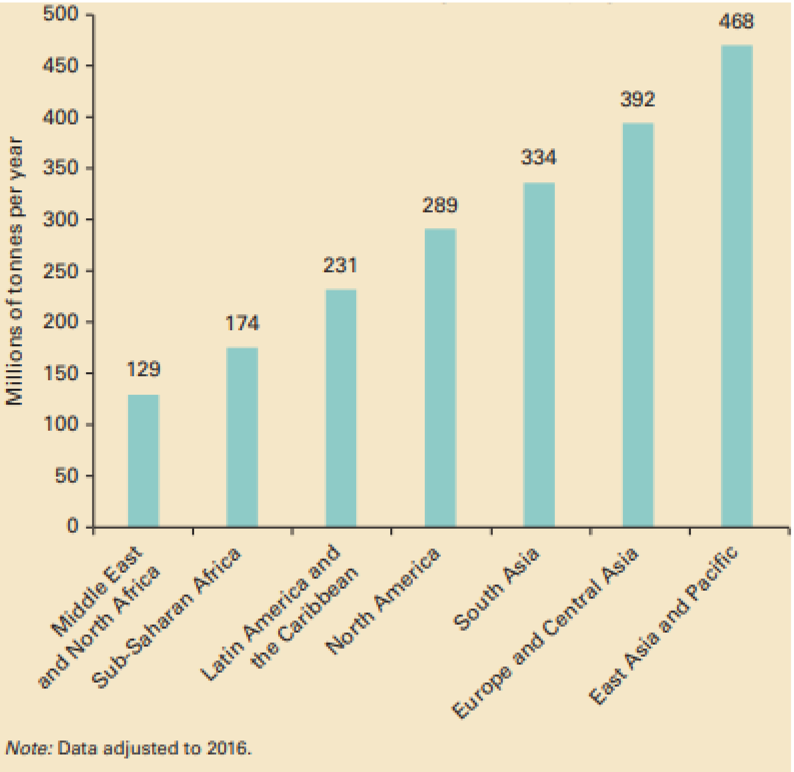


Figure 2.2 - 18: Amount of waste generated by Region (Kaza *et al.*, 2018; The world bank, 2020)

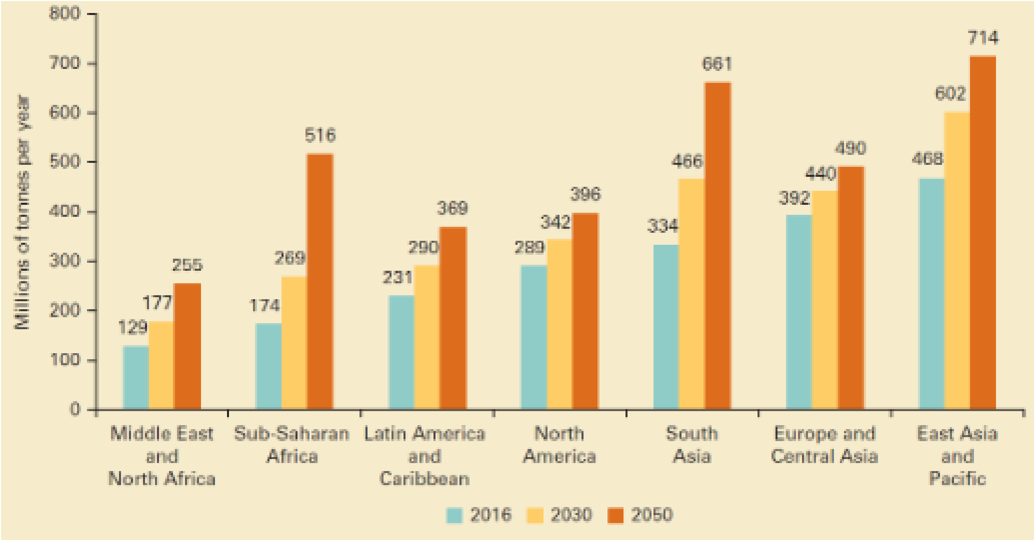


Figure 2.2 - 19: Waste generation projection (Kaza *et al.*, 2018; The world bank, 2020)

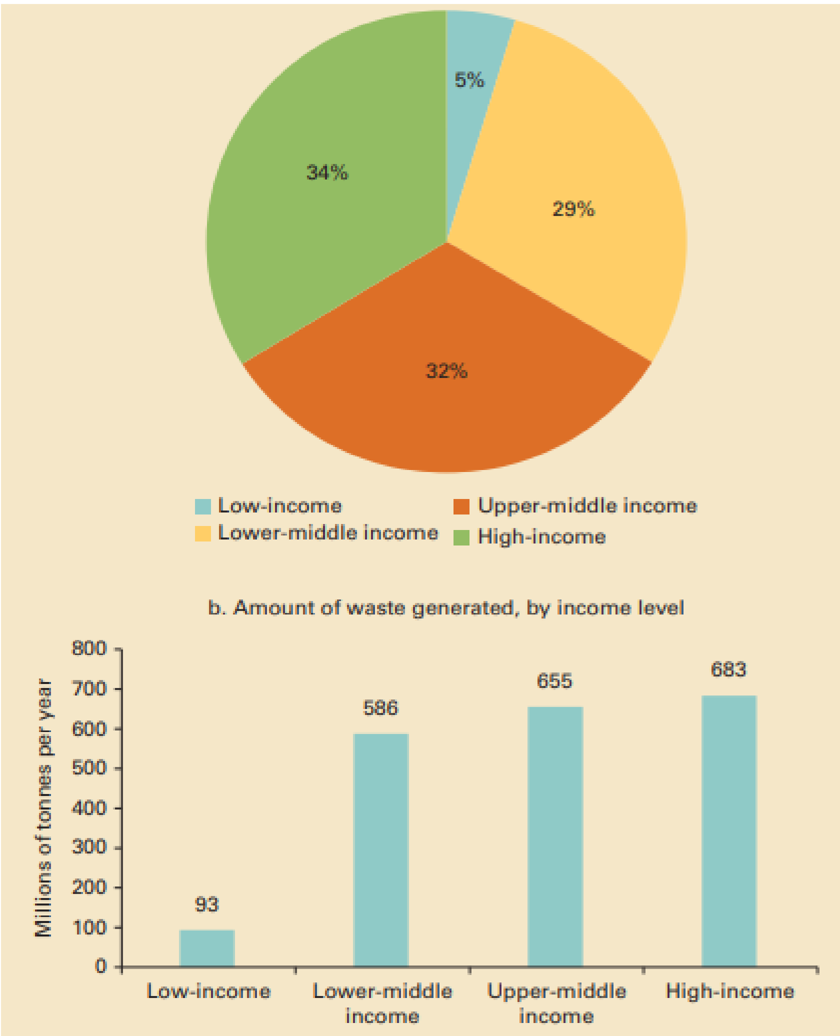


Figure 2.2 - 20: Amount of waste generated, by income level for 2016 (Kaza *et al.*, 2018)

Global waste composition varies significantly amongst different income level countries. This is motivated by the varying habits and lifestyles of individuals in those countries, as well as the availability of resources. In low and middle-income countries, food and green waste constitute about 50% of waste, whilst in high-income countries, a comparable amount of organic waste is generated (Kaza *et al.*, 2018). The global waste composition for foods and greens is 12%. (Figure 2.2-21). The highest composition is that of plastic waste with 44%. Waste collection is the most important stage in waste management. It is arguably the city's largest budget item in the management of solid waste (Polasi, Matinise and Oelofse, 2020). It is at this stage that waste sorting, separation and characterisation is made possible, making it economically feasible to perform the latter stages in the handling of waste such as waste treatment and disposal. Waste collection rates varies between income levels, as high and upper-middle-income countries have access to global waste collection systems whilst waste collection rates varies between urban and rural areas in low-income level countries (Kaza *et al.*, 2018).

In low-income countries about 48% of waste is collected in cities but only about 26% of total waste is collected in rural areas, whereas in middle-income countries, rural waste collection varies between 33-45% (Kaza *et al.*, 2018).

Waste disposal is another vital step that needs to be handled with utmost care. The trends in waste disposal vary between income level countries. On a global level, 37% of waste is disposed of in some form in the landfill, 33% is open dumped, 11% undergoes incineration, and 19% of the waste is recycled or composted (Kaza *et al.*, 2018) (Figure 2.2-22). With regards to the handling of waste, more than one-third of waste in high-income countries is recycled and/or composted, whereas in middle to low-income countries, a majority of the waste is landfilled (Kaza *et al.*, 2018).

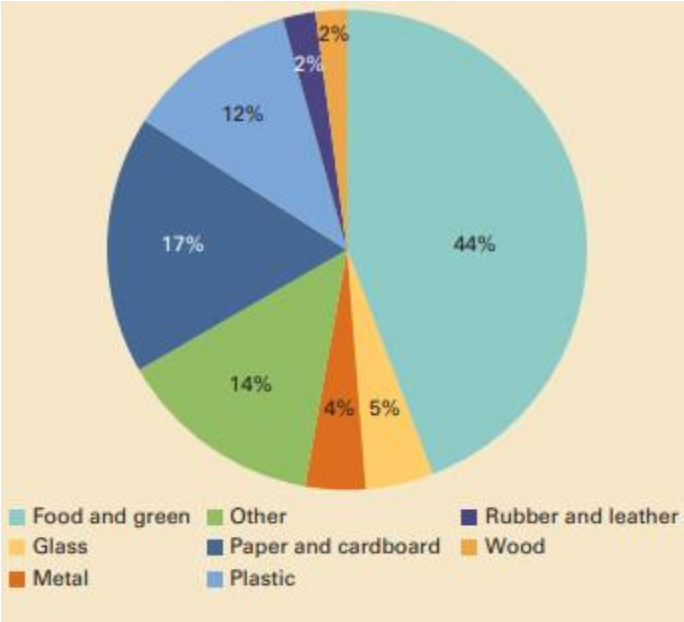


Figure 2.2 - 21: Waste Composition, globally (percentage) (Kaza *et al.*, 2018)

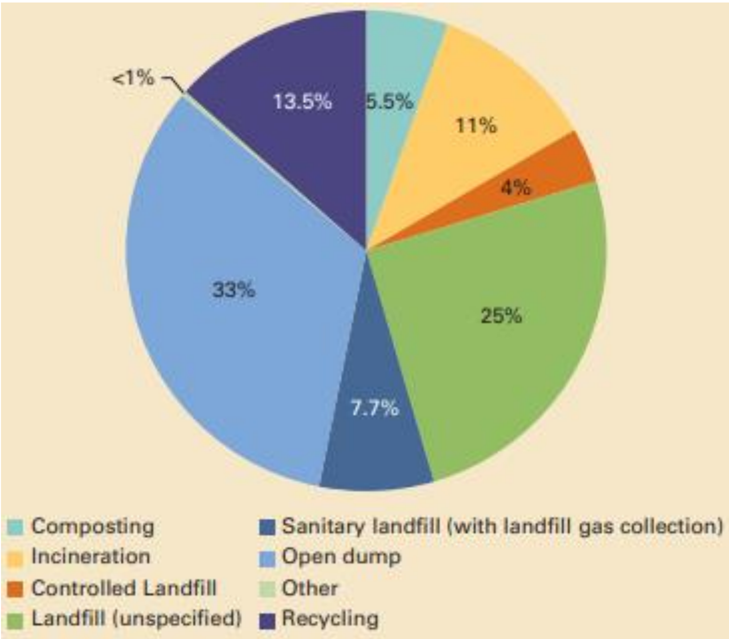


Figure 2.2 - 22: Waste treatment and disposal (percentage) (Kaza *et al.*, 2018)

2.2.6 Waste disposal technologies explained

This section presents the various waste disposal technologies available to developing countries such as South Africa.

Landfilling

The most frequently used waste disposal method used in South Africa is landfilling, as 90% of all waste generated is taken to landfill sites (Nkosi, 2014). In waste management, landfilling is a traditional method of waste disposal where waste is simply disposed of in designated areas and allowed to disintegrate. ‘‘A modern landfill is an engineered method for depositing waste in specially constructed and protected cells on the land surface or in excavations into the land surface’’(Danthurebandara *et al.*, 2013). Of all the other methods to be discussed in this section, this is the cheapest and easiest way to dispose of waste. However, landfilling poses several health risks that arise from the microbial activity on waste. During the degradation of waste, gases (predominantly methane) and liquids (leachate) are produced. 20% of the global anthropogenic methane emission comes from landfills (Danthurebandara *et al.*, 2013). Leachate is known to contaminate surface and underground water. Other problems include wind dispersing debris, rodent, insect and bird infestation, as well as being a spontaneous combustion hazard (Muzenda, 2014).

Composting

Composting is a naturally occurring process that aerobically decomposes organic matter producing a stable product called humus (Argun *et al.*, 2017). During the composting process, steam is produced, carbon dioxide and heat are released into the atmosphere, and compost is converted to humus (Argun *et al.*, 2017). Compost is not a fertilizer, but it can be used to enrich the soil, thereby assisting with moisture retention and controlling plant disease (Argun *et al.*, 2017).

Incineration

Incineration is a thermal waste to energy (W-t-E) technology in which organic materials are combusted at temperatures in excess of 850 (Tawona, 2015). The heat energy released from this process, depending on the setup, can then be used to generate electricity. This method is useful in crowded cities to reduce the mass and volume of waste taken to the landfill (Patil and Amol A.Kulkarni, 2014). Using this method, up to 90% volume of waste and 75% mass is reduced (Masebinu *et al.*, 2016). The ash produced from incineration can be used in environmentally friendly constructions (Patil, Kulkarni and Patil, 2014). Some issues with incineration are the cost associated with constructing and operating an incinerator; as well as the required skilled personnel for the complex incineration process (Patil, Kulkarni and Patil, 2014).

Pyrolysis

Pyrolysis is the thermal degradation of lignocellulosic derivatives under inert conditions in an oxygen-deficient environment (Zaman *et al.*, 2017). The end products of pyrolysis are char, non-condensable gases and bio-oil. Common emissions include methane, hydrogen, carbon monoxide and carbon dioxide (Zaman *et al.*, 2017). During pyrolysis, larger organic molecules are broken down into smaller molecules which are then released as either gases, condensable vapours (tar and oils) or as solid char (Masebinu *et al.*, 2016). The composition of the product stream is influenced by temperature, time, heating rate, pressure, reactor design and configuration (Zaman *et al.*, 2017).

Gasification

Gasification is the process by which carbon-containing feedstock is converted to synthesis gas (syngas). This process takes place at elevated temperatures, and the heating of feedstock is done indirectly. Syngas is composed of carbon monoxide (CO), carbon dioxide (CO_2) and hydrogen (H_2) gases. Char and tar are also the products of the gasification process. The by-products include hydrogen sulphide (H_2S), and slag (minerals from coal) (Breault, 2010). Syngas can be used as a fuel to generate electricity, and it also finds great usage in the petrochemical and refining industries for the production of hydrogen (Breault, 2010). Potential feedstocks for gasification include organic matter, wood, biomass and plastic (Tawona, 2015).

Anaerobic digestion

Anaerobic digestion (AD) is a naturally occurring process. During this process, organic matter is broken down in the absence of oxygen, and biogas is released, leaving behind an organic residue called digestate. Biogas is comprised of methane, carbon dioxide and water. A typical compositional breakdown of biogas is presented in Table 2.2-7. Biogas is further processed and used to produce electricity and heat or rather as a natural gas substitute. The digestate is nutrient-rich and can be further processed and used as a fertiliser to improve soil fertility. This process is carried out particularly on landfill sites or wherever waste can be found buried under a heap, creating a warm and oxygen-deficient environment. These conditions make it suitable for microorganisms to degrade the waste. The microbial activity, in the absence of oxygen, on organic waste results in the release of environmentally harmful methane gas into the atmosphere. However, when methane is captured, and further processed, it can be put to good use as a fuel thereby reducing reliance on fossil fuels. This requires executing the process in a specially designed AD reactor (digester) from which the gas can be recovered, and the by-products withdrawn for further treatment and usage. AD is considered one of the most viable options for municipal solid waste management (Tawona, 2015).

Table 2.2 - 7: Typical Composition of Biogas from Organic Fraction of Municipal Solid Waste (Greben and Oelofse, 2009)

Components	Symbol	Concentration (vol. %)
Methane	CH_4	55–60 (50–75)
Carbon dioxide	CO_2	35–40 (25–45)
Water	H_2O	2 (20 °C)–7 (40 °C)
Hydrogen sulfide	H_2S	20–20 000 ppm (2%)
Nitrogen	N_2	< 2
Oxygen	O_2	< 2
Hydrogen	H_2	< 1

2.3 Assessing Mechanical Biological Pre-Treatment (MBT) prior to landfilling

Prior to 2001, there was no example of organic waste management which was conducted in Durban, South Africa. All organic waste was sent to landfill. MBT was the first technology that was implemented in 2001, at an experimental stage which ran between 2001 and 2010. This was the first technology assessed for sustainable organic waste treatment. The technology never took root and was not upscaled due to various limitations. The following section presents the results and limitations of that experimentation.

2.3.1 Mechanical Biological Treatment Facility in South Africa

This section discusses the instalment and operation of a Mechanical Biological Treatment (MBT) facility that was set up in Durban, South Africa in 2002 as an attempt to investigate its feasibility on a developing country like South Africa. This was a first of its kind in South Africa and was brought to fruition by the productive collaboration between the University of KwaZulu Natal, Tshwane University of Technology as well as the Technical University of Dresden. The setup was done in comparison to an already operating setup located in Germany known as the Dome Aeration windrow Technology (DAT). There hasn't been a landfill wide implementation of the technology. However, various pilot research on different parameters has been conducted to date (Griffith and Trois, 2006; Trois and Polster, 2006; Trois et al., 2007; Trois and Griffith, 2008; Griffith, 2009).

MBT technology is a very promising approach to stabilising waste before landfilling, especially in developing countries like South Africa, where legislation on waste management is heavily reliant on concentrating and containing of waste without separation at source (Trois et al., 2007). The large presence of organic waste and high moisture waste has led to increased biological activity in the landfills, resulting in release of environmentally "unsafe" gases, and contamination of air and underground water systems. Mechanical, physical separation of waste coupled with a biological process such as Anaerobic Digestion can provide a solution for treatment of waste at landfills. It was stated that theoretically, 98% of emissions at landfill can be avoided (Trois and Couth, 2012).

This setup was placed at Bisasar landfill, which is also home to a more sophisticated landfill gas-to-energy facility (eThekwini Municipality). Landfill gas-to-energy is a technology that harnesses and diverts gasses released by the waste but does not stabilise the waste. Bisasar landfill started its operation in 1980 and it has been managed by Durban Solid Waste (DSW) since 1995. It is the oldest operating landfill in South Africa (Trois et al., 2007) and the largest in Africa (eThekwini Municipality).

2.3.2 Mechanical Biological Treatment setup at Bisasar road landfill

Durban became the first city in South Africa to host a Mechanical Biological Treatment setup at its Bisasar road landfill. In a paper titled "Introducing mechanical biological waste treatment in South Africa: A comparative study", Trois et al., (2007) indicated that the reason for selecting this area was because Durban is representative of waste management in a typical South African metropolitan area. The MBT setup as already stated utilized the Dome Aeration Technology (DAT) which is used in Germany and it was selected because of its affordability to a developing country such as South Africa in terms of capital cost, energy input, simple design and as well a potential for employment (Trois et al., 2007). The operation of a Dome Aeration system is such that the aeration process within a windrow of waste is driven by thermal convection (Mollekopf et al., 2002 cited in Trois *et al*., 2007). The operating principle of a DAT windrow system is shown in Figure 2.3-1 below. The thermal convection is achieved through creation of large voids for flow of gases, named Domes and Channels (Figure 2.3-2). The channels let in ambient air which then facilitates the degradation reactions, producing gases which are then driven through smaller voids in the waste to the Domes and out through the chimneys. Smaller voids were created by mixing in structural waste in the windrow (Griffith and Trois, 2006). Structural waste was composed of dry branches and planks. Green branches were rejected.

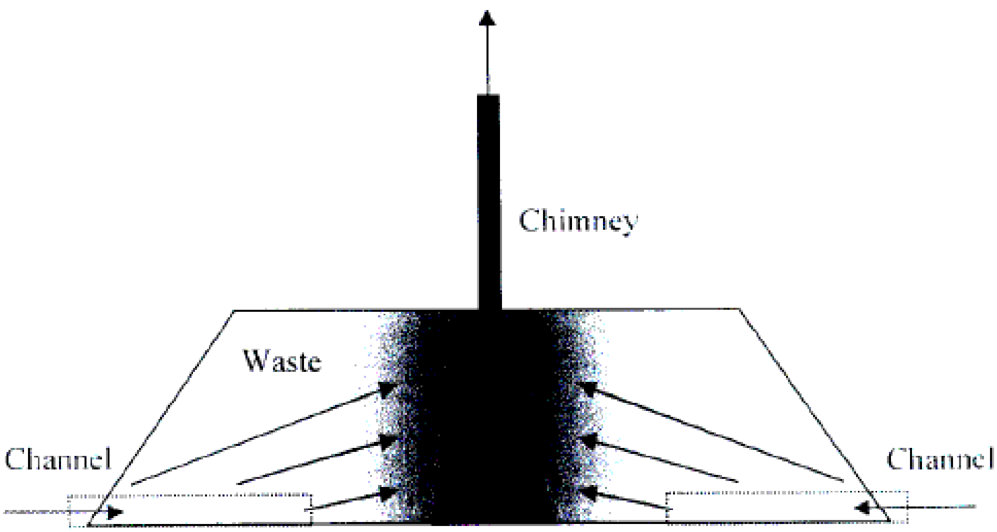


Figure 2.3 - 1: Operating principle of a DAT windrow system (Griffith and Trois, 2006)

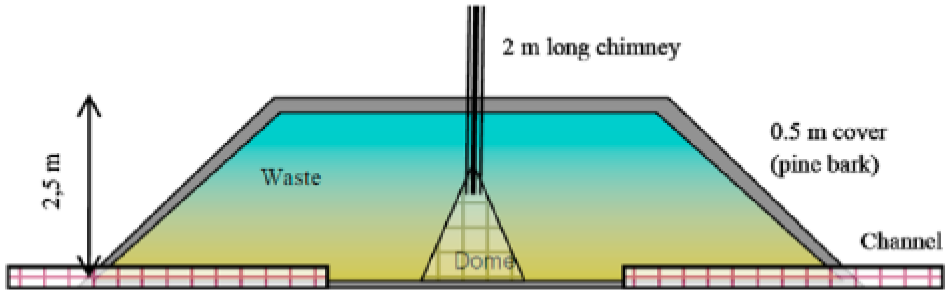


Figure 2.3 - 1: Cross sectional view (A-A) of a windrow constructed in (Trois and Simelane, 2010)

2.3.3 Passively Aerated DAT Windrow setup for the pilot at Bisasar landfill

In assessing the potential for a Mechanical Biological Treatment facility in South Africa, utilising the Dome Aeration windrow treatment technology, issues of climate, input material as well as availability of utilities and facilities were raised. In the vast research performed on the subject at Bisasar landfill in Durban, representative of a typical metropolitan landfill and waste stream in the country, passively aerated windrow treatment was found to successfully stabilise the waste bound for landfilling. The results which were produced and comparable to the output registered in Germany proved that the Green House Gas (GHG) pollution potential of waste is massively reduced compared to current trends of untreated waste. Mechanical treatment such as shredding, and sorting were found to be detrimental to the success of the biological treatment in terms of removal of pollutants such as NH_3-N. Furthermore, landfill space saving through volume reduction of waste is achieved. However, various recommendations were drafted from the numerous tests conducted (Trois et al., 2007; Griffith, 2009; Trois and Simelane, 2010; Couth and Trois, 2012):

- Source sorting of waste is important to improve the efficiency of the technology
- Recycling of slowly degrading waste materials which are found not to be sufficiently treated by the windrow system was recommended as they are a risk for a long-term pollution problem
- Legislation must be introduced that support mechanical and biological treatment of waste prior to landfilling and diversion of garden waste and recyclable waste
- Windrow processing if conducted through initiatives such as the Clean Development Mechanisms can be financially viable.

Because windrow process is merely a pre-treatment step to stabilize the waste bound for landfilling, where the conditions are mostly anaerobic, a test of how the waste behaved post-treatment (where it is mostly anaerobic) was conducted (Trois and Griffith, 2008; Griffith and Trois, 2006; Trois and Simelane, 2010; Trois and Polster, 2006). A pilot setup was erected and fed with output material from windrows and the resulting material characterized. The degradability was measured as reduction in pollution parameters such as Chemical Oxygen Demand (COD), Biological Oxygen Demand (BOD), Ammoniacal Nitrogen (NH3-N) as well as Volatile Solids (VS) and Total Solids (TS).

Investigations for 8 weeks long windrow treatments showed a drop in COD values up to 78% as presented in Table 2.3-1. BOD_5 reduction varied from 35% to 81% for 8 weeks treatments, in the various treatments performed. Trois et al. (2007) reported that the final BOD dropped from the initial 2711 mg/l to 529 mg/l in 8 weeks and an even further 469 mg/l in 20 weeks. For the same experiment, (NH_3-N) was dropped from 84.3 mg/l to 1.14 mg/l in 20 weeks. This represented a 99% elimination of this pollutant. VS loading in the waste stream was dropped by 35-37%. An investigation performed on sorted waste fractions showed that course input received a far better treatment than fine waste. A drop of 80% in COD values for fine materials was measured at the end of the composting as compared to 81% for coarse material. A 91% reduction in volatile solids for coarse material was observed in comparison to 84% from fine material. Final results from a 16-week compost were found to be higher than that from 8 weeks. This was attributed to the inefficiency of the process (Trois and Griffith, 2008).

The assessment of MBT (aerobic) under anaerobic conditions showed a removal of acidic inhibition which is normally defined for untreated waste (Trois and Griffith, 2008). The success in removal of acidic inhibition from the waste shows that the waste can be further treated anaerobically for energy recovery and other valuable end products. This was mainly for the finely separated particle sized compost. Coarse material showed some inhibition which may have been due to a high porosity and thus they performed better aerobically as displayed in Table 2.3-2 (Trois and Griffith, 2008). These results showed that MBT alone is not enough to combat the pollution problem at the landfills. The waste situation can only be dealt with from collection through physical separation of waste. Coarse waste is more suitable for composting whilst fine organic material requires Anaerobic Digestion. The long-term COD release if a suitable waste treatment is not used equals that of untreated waste.

Table 2.3 - 1: Reduction in various performance parameters to untreated waste for 8 weeks

Parameter	(Trois and Simelane, 2010)	(Couth and Trois, 2012)	(Griffith, 2009)	(Trois et al., 2007)	(Griffith and Trois, 2006)
COD %	22.9		70	71 (69)	78
BOD_5 %	35.7	50%		80 (83)	81
BOD_5/COD%	16.7				15.5
VS %	35.9			36 (29)	37
NH_3-N %	Increase		50	79 (99)	75
pH %	Increase			Increase	22
Maximum Temperature	65	70	70	70	70

NOTE: The values in brackets refer to >8 weeks windrow tests.

Table 2.3 - 2: Reduction in pollution parameters in Sorted waste test (8 weeks) (Trois and Griffith, 2008)

Parameter	Fine	Coarse
COD %	80 (54)	81 (52)
VS %	84 (63)	91 (61)
NH_3-N %	70 (44)	78 (52)

NOTE: The value in brackets refer to 16-week windrow test

<u>Final criticism/lessons learnt</u>

Despite the lengthy experimentation on the technology, it was never upscaled to cover a landfill wide operation. The results showed that MBT alone cannot be used to stabilise the waste at a landfill. Due to the associated costs in the instalment of the technology such as energy costs and heavy machinery, and the lack of renumeration to recover the finances, this process was thus not feasible. Lastly there was so much CO_2 emissions from the windrows which makes an upscale a pollution risk.

2.4 Focus on Anaerobic Digestion (AD)

The analysis of the results from the experimentation on MBT showed that separate waste collection and MBT can be effectively used as pre-treatment technologies for AD. This section provides an in-depth exploration of the AD technology.

2.4.1 Overview of Anaerobic Digestion

Anaerobic digestion of waste in a controlled environment (or specially designed reactor) is not a novel invention. Its origin dates back to the 17^{th} century, to a curious mind of the scientist, Van Helmont. In the year;

1630: Jan Baptista Van Helmont records seeing "the emanation of flammable gases from decaying organic matter" (Gunnerson and Stuckey, 1986; Verma, 2002). This was not the first time this emergence of gases was observed. Previously described as the emergence of flickering lights and flames, this occurrence was at the time mistakenly associated with the existence of dragons (Gunnerson and Stuckey, 1986).

1667: A more detailed description of this gas was provided by Shirley, who is often referred to as the discoverer (Gunnerson and Stuckey, 1986).

1776: Count Alessandro Volta was credited for scientifically explaining methane digestion (Gunnerson and Stuckey, 1986). He noted that the amount of gas that evolves is a function of the amount of organic matter from which it emerges, secondly, a portion of the gas emitted forms an explosive mixture with air (Gunnerson and Stuckey, 1986; Verma, 2002). This explains the first observation of flickering lights being confused with the existence of dragons.

1804: Dalton establishes and records the chemical composition of methane gas.

1808: Sir Humphry Davy demonstrates that methane can be produced from anaerobic digestion of cattle manure; from this moment organic waste was recognised as a source of energy.

1859: In this year the application of anaerobic digestion at an industrial level was launched. The first digestion plant was built in Bombay, India (Verma, 2002; PennState Extension, 2012).

1884: Gayon, a student at the time, fermented manure at 35°C and obtained 100 litres of methane per cubic meter of manure (Gunnerson and Stuckey, 1986). From this research, it was established that organic waste fermentation could be used for heating and lighting.

1896: Recorded as 1895 in some publications, streetlights in Exeter, England were powered from biogas obtained from a “carefully designed” sewage waste treatment facility (Gunnerson and Stuckey, 1986; Verma, 2002; PennState Extension, 2012)

The 1920s-1930s: Buswell publishes information on anaerobic bacteria and the ideal conditions for culturing (Gunnerson and Stuckey, 1986; Verma, 2002; PennState Extension, 2012).

The AD technology for fuel production has not received much attention due to the costs associated with it as compared to cheaper alternatives such as coal and petroleum fuels. There was, however, a peak usage during World War 2 when fuel demand was high (Verma, 2002).

<u>Trends in anaerobic digestion technology utilization</u>

Small scale AD systems were the first to find ground for energy generation in most countries, especially in rural areas. During the energy crisis in 1973 and 1979, countries such as India, China and Southeast Asia increased reliance on AD systems (Verma, 2002). Small size digesters were developed for production of biogas to power villages, and were fed with human, animal and kitchen wastes (Verma, 2002). Despite the considerable research done on the subject, in developed countries, a limited understanding of AD technology at an industrial level remained. Countries such as China, India and Thailand reported up to 50% failure rates and up to 80% in the United States of America for farm-based digesters (Verma, 2002). This, however, did not deter countries from further developing this technology. About 6 to 8 million family-sized, low technology digesters, located mainly in farm-based facilities, are used to provide biogas for cooking and lighting fuels (Verma, 2002; PennState Extension, 2012).

Prior to 2012, Europe had built more than 600 farm-based AD digesters in 5 years, with 250 new systems installed in Germany alone (PennState Extension, 2012). AD systems have a good record of treating a wide range of waste streams found in Europe, such as farm, industrial and municipal wastes (Verma, 2002; PennState Extension, 2012). The country most invested in this system in Europe is Denmark. By 2012, Denmark had established over 18 large and centralized AD plants which used a feedstock of co-digested manure, clean organic industrial wastes and source-separated municipal solid waste (MSW) (PennState Extension, 2012).

AD technology has also been used extensively in the treatment of industrial wastewaters. In 2012, more than 1000 vendor-supplied AD systems were reported to be operational, or under construction., of which 44% were installed in Europe, 14% in North America and the remainder located in Brazil, where they are used in the treatment of vinasse co-product from sugarcane-based ethanol production (Verma, 2002). Industries that use this technology include pharmaceuticals, fibre, food, meat, milk etc. Most industries use AD as a pre-treatment step in order to lower sludge disposal costs, control odours and reduce costs associated with municipal wastewater treatment (PennState Extension, 2012). The first AD digester that utilized municipal solid waste was in the United States of America from 1939-1974 (PennState Extension, 2012).

Relevance in the present age

There is extensive research on the AD technology for usage in solid waste as well as wastewater sludge. This includes major break throughs such as that of Buswell in the 1930s that defined the optimal conditions for increased methane yield. The advancements in operational understanding and acknowledgement of the benefits of this process, new procedures, equipment, and closed reactor designs were developed and implemented. Both developing and developed countries around the world are striving towards green energy and moving away from fossil fuels. This is mainly due to the fact that the natural resources are being depleted and the negative impact that fossil-based fuels have on the environment. Furthermore, nations are targeting alternatives municipal solid waste accumulation other than disposal. In the near future, some developing and developed countries intend to ban landfilling altogether, while others, aim to limit the amount of organic waste taken to the landfill.

Anaerobic Digestion as a waste management solution in the Biorefinery Industry

The biorefinery industry deals with the valorisation of waste materials for economic gain. The need for sustainability, safety and economic growth prompted the rise of various technologies that aim to respond to the economic needs by creating a bioeconomy which utilises the available bio-based-resources (Hingsamer and Jungmeier, 2019). Anaerobic digestion is one of the most important biorefinery technologies that seeks to create a bioeconomy through recycling of food waste. This technology is an important step toward complete circularisation of the food resource and at the same time creating a sustainable waste management solution that can be relied upon by future generations.

2.4.2 The Basics of Anaerobic Digestion

The process of anaerobic digestion proceeds in four stages, all facilitated by different groups of microorganisms, namely, i) hydrolysis, ii) acidogenesis, iii) acetogenesis/dehydrogenation, and iv) methanogenesis (Weiland, 2010). These microorganisms place different requirements on the environment on which they operate, thus requiring a syntrophic relationship (Jingura and Kamusoko, 2017b). The first and second as well as the third and fourth groups of bacteria are closely linked to each other and therefore, the process can be achieved in two stages (Weiland, 2010). The different activities occurring in these stages are outlined below.

i) Hydrolysis

During hydrolysis/liquefaction, large complex insoluble organic matter is broken down into smaller constituents (monomers) which are soluble (Sawyerr *et al.*, 2019). These large often insoluble macro-molecules, as shown in Figure 2.4-1 include carbohydrates, fats and proteins which are broken down into simple sugars, fatty acids and amino acids, respectively (Zupancic and Grilc, 2012). Carbohydrates can be degraded within hours, however cellulose, proteins, and fats require a few days to disintegrate (Weiland, 2010). Hydrolysis is a very important step in the anaerobic digestion process and is often rate-limiting, thus in industrial applications, the process is at times enhanced using chemical reagents (Verma, 2002). The rate at which hydrolysis occurs depends on factors such as particle size, production of enzymes, and the diffusion and adsorption of enzymes on the substrates (Shah *et al.*, 2017). The hydrolysis reactions in AD can be biological (using hydrolytic microorganisms), bio-chemical (using extracellular enzymes), chemical (using a catalyst) and/or physical (by means of thermal energy and pressure) in nature (Zupancic and Grilc, 2012). Hydrolysis reactions are summarised below:

Lipids → fatty acids

Polysaccharides → monosaccharides

Protein → amino acids and peptides (amino acids can be utilized as a source of energy by anaerobic microorganisms (Shah *et al.*, 2017)

Nucleic Acids → purines and pyrimidines

A cellulose hydrolysis reaction can be written as follows:

$$(C_6H_{10}O_5)n + n\,H_2O \rightarrow \; n\,C_6H_{12}O_6 + n\,H_2 \qquad \text{Equation 2.4-1}$$

The reaction follows the hydrolysis of cellulose through the addition of water to form glucose as the main product and the release of hydrogen. This reaction is catalyzed by homogenous and heterogeneous acids to yield a fermentable monosaccharide (glucose), which can then be acted upon by bacteria (Anukam *et al.*, 2019). Some of the products from hydrolysis such as hydrogen (H_2) and acetates (CH_3COO^-) can be directly utilized by methanogens, whilst others, with relatively larger molecules are dealt with in the next step of the AD process (Zupancic and Grilc, 2012; Anukam *et al.*, 2019). Hydrolytic enzymes such as cellulases, xylanases, amylases, lipases and proteases are secreted by hydrolytic bacteria (Shah *et al.*, 2017). These facultative and strict anaerobic bacteria include *Clostridia*, Bacteroides, Bifidobacteria, *Streptococci* and *Enterobacteriaceae* (Weiland, 2010).

ii) Acidogenesis

In this stage, acidogenic (fermentative) bacteria further degrade soluble compounds including products from the hydrolysis stage and convert them to carbon dioxide, hydrogen and acetic acid (Zupancic and Grilc, 2012; Anukam *et al.*, 2019). Acetic acid is the most desirable product from this stage as it is used as a substrate by methane producing microorganisms (Anukam *et al.*, 2019). Other products include volatile fatty acids (VFAs) which are favoured at pH >5 and ethanol which is characteristic at pH <5 (Anukam *et al.*, 2019). A high concentration of hydrogen is formed by acidogenic microorganisms, usually the fastest step in a balanced anaerobic process, however, for the process phase to be thermodynamically favourable, the hydrogen concentration must be kept low (Sawyerr *et al.*, 2019). The reactions taking place are shown below:

$$C_6H_{12}O_6 \leftrightarrow 2\,CH_3CH_2OH + 2\,CO_2$$ Equation 2.4-2

$$C_6H_{12}O_6 + 2\,H_2 \leftrightarrow 2\,CH_3CH_2COOH + 2\,H_2O$$ Equation 2.4-3

$$C_6H_{12}O_6 \rightarrow 3\,CH_3COOH$$ Equation 2.4-4

iii) Acetogenesis

At this stage simple molecules, volatile fatty acids and alcohols from acidogenesis are broken down by acetogens into mostly acetic acid as well as carbon dioxide and hydrogen (Shah *et al.*, 2017). Despite producing hydrogen, the metabolism of acetogenic bacteria is inhibited by hydrogen accumulation (Tawona, 2015; Anukam *et al.*, 2019). This is a problem that is often exacerbated by excess hydrogen from acidogenesis, and therefore relies on methanogenic bacteria to utilize the hydrogen effectively to detoxify the process (Anukam *et al.*, 2019). It is important to keep the partial pressure of hydrogen low. However, Weiland (2010) reported that hydrogen may be a limiting substrate for methane producing microbes.

This may be due to the presence of hydrogen producing bacteria in the natural biogas consortium increasing the daily biogas yield. Homoacetogenic bacteria, such as *Acetobacterium woodii* and *Clostridium aceticum*, are responsible for converting hydrogen and carbon dioxide to acetate(Weiland, 2010; Shah *et al.*, 2017). During acetogenesis, 25% of acetates are formed and 11% of hydrogen released. Since 70% of methane arises from reduction of acetates it therefore holds true that this phase represents the efficiency of the biogas production (Shah *et al.*, 2017). The reaction series from this stage is as follows:

$$CH_3CH_2COO^- + 3\,H_2O \leftrightarrow CH_3COO^- + H^+HCO_3{}^- + 3\,H_2$$

Equation 2.4-5

$$C_6H_{12}O_6 + 2\,H_2O \leftrightarrow 2\,CH_3COOH + 2\,CO_2 + 4\,H_2$$

Equation 2.4-6

$$CH_3CH_2OH + 2\,H_2O \leftrightarrow CH_3COO^- + 3\,H_2 + H^+$$

Equation 2.4-7

iv) Methanogenesis

This is the final stage of the AD process and the methane forming phase. This is often the rate-limiting stage, depending on methanogens growth rate. At this stage methanogenic bacteria convert acetic acid or hydrogen and carbon dioxide to methane (Weiland, 2010). This is achieved in two ways; either by cleavage of acetic acid molecules to generate carbon dioxide and methane or by reducing carbon dioxide with hydrogen to produce methane and water (Anukam *et al.*, 2019). The latter has a high rate of methane production. Only selected methanogens such as *Methanosarcina barkeri*, *Metanonococcus mazei*, and *Methanotrix soehngeni*, are able to act on acetates, whereas all methanogenic bacteria are able to use hydrogen (Weiland, 2010). However, limited hydrogen concentrations result in the acetate reaction being the main producer of methane (Verma, 2002). This stage is sensitive to both high and low pH and is limited to a pH between 6.5 and 8.0 (Weiland, 2010; Shah *et al.*, 2017; Anukam *et al.*, 2019). The methanogenesis reaction series is shown below:

$$CH_3COOH \rightarrow CH_4 + CO_2$$

Equation 2.4-8

$$CO_2 + 4\,H_2 \rightarrow CH_4 + 2\,H_2O$$

Equation 2.4-9

$$2\,CH_3CH_2OH + CO_2 \rightarrow CH_4 + 2\,CH_3COOH$$

Equation 2.4-10

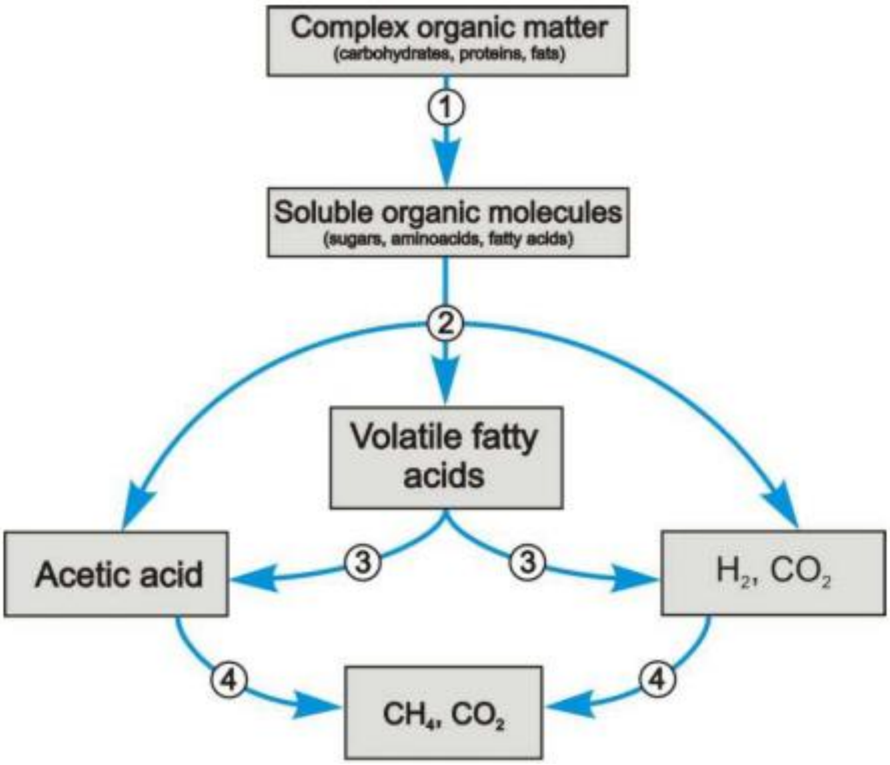


Figure 2.4 - 1: The anaerobic digestion pathway of organic matter (Zupancic and Grilc, 2012)

2.4.4 The Process of Anaerobic Digestion

The process of anaerobic digestion generally involves pre-treatment, waste digestion, gas recovery and residue/digestate treatment (Verma, 2002). A general process flow for anaerobic digestion with co-digestion of feed is shown in Figure 2.4-2. The process begins with a pre-treatment stage where the bio-waste is prepared for microbial digestion (Kigozi, Aboyade and Muzenda, 2014). The pre-treatment varies according to feedstock and can involve shredding and sorting of Municipal Solid Waste (MSW) or mixing to ensure homogeneity. Often source separation of waste is done, and non-digestible and recyclable waste such as plastics are removed. A homogenous feed is then fed into the digester (Masebinu *et al.*, 2018).

Apart from the physical pre-treatment already discussed, chemical and biological pre-treatment techniques are also used. Chemical pre-treatment is used for treating lignocellulosic material such as spent grains and silage (Zupancic and Grilc, 2012). Chemical treatment is commonly used in combination with heat, pressure, and the application of either an acid (e.g., hydrochloric acid) or alkaline solution (e.g., sodium hydroxide) (Zupancic and Grilc, 2012). Biological pre-treatment involves degradation of organic material using enzymes (Zupancic and Grilc, 2012).

The digestion unit is the most important component in the AD process (Masebinu *et al.*, 2018). Inside the digester, the contents are diluted to meet the required solid content. The dilution can be done using sewage water, clean municipal water or liquid effluent from the digester can be recirculated. A working temperature setting is ensured inside the digester, typically by use of a heat exchanger. The contents are retained in the digester until all the biogas is collected. Afterwards, the biogas is further treated to improve methane purity. To achieve the required purity, water scrubbing can be employed, and carbon dioxide removed (Zupancic and Grilc, 2012). It is difficult to determine which digester is suitable for the organic waste, therefore it is important to conduct laboratory and pilot-scale experiments on the waste before a large digester is employed (Zupancic and Grilc, 2012).

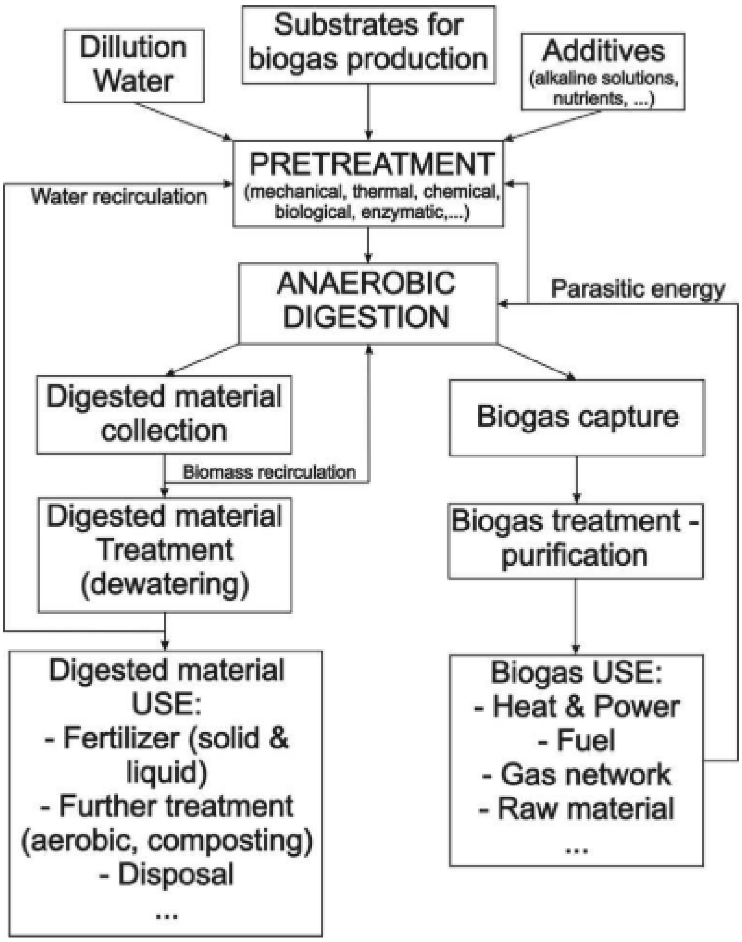


Figure 2.4 - 2: Anaerobic digestion block flow diagram and biogas utilisation (Zupancic and Grilc, 2012)

2.4.5 Important Parameters in Anaerobic Digestion

Anaerobic digestion is a very challenging process. A large amount of capital is spent from the pre-treatment stage to gas upgrading. This highlights the importance of operating under optimal conditions. Optimal operation ensures the best conditions for microbial activity whilst weighing the capital investment. The idea is to find a balance between these parameters to render the process optimal, effective, and feasible. For instance, conditions required by acidogenic bacteria differs from that of methanogenic bacteria. If one digester is used where all AD processes take place sequentially, meeting the requirements for methanogenic bacteria will take preference as they tend to have longer regeneration times (Table 2.4-1), slow growth rates and are more sensitive to environmental conditions (Zupancic and Grilc, 2012). This makes methanogenesis a rate-limiting step, however, for cellulose-containing substrates, hydrolysis is rate-limiting and for fat-rich substrates, acetogenesis is rate-limiting. Therefore, special consideration must be given to these stages (Zupancic and Grilc, 2012).

Table 2.4 - 1: Time of Regeneration of microorganisms (Zupancic and Grilc, 2012)

Microorganisms	**Time of regeneration**
Acidogenic bacteria	Less than 36 hours
Acetogenic bacteria	80-90 hours
Methanogenic archaea	5-16 days
Aerobic microorganisms	1-5 hours

Some of the vital parameters affecting the performance of an AD process is discussed below. These parameters are influenced by feedstock characteristics, reactor design and operational conditions (Weiland, 2010; Jingura and Kamusoko, 2017a). Feedstock characteristics include feedstock composition, Volatile Solids (VS), Chemical Oxygen Demand (COD), C/N ratio, as well as presence of inhibitory substances (Babaee and Shayegan, 2011). Temperature and retention time are examples of operational conditions.

i) Temperature

There are three established operating temperature ranges that provide the best environment for microorganisms. These are the psychrophilic (15-20^{o}C), mesophilic (30-40^{o}C) and thermophilic (50-60^{o}C) ranges (Zupancic and Grilc, 2012; Jingura and Kamusoko, 2017b). The most common are mesophilic and thermophilic (Verma, 2002). Microbial activity in the AD process increases with temperature. It has been reported that the rate of digestion during a thermophilic process can be four times higher than that of a mesophilic digestion (Zupancic and Grilc, 2012). Therefore, thermophilic digestion has shorter residence times as compared to mesophilic digestion. It has been reported that gas production ceases when ambient temperatures drops below 10°C (Gumisiriza *et al.*, 2017).

Thermophilic bacteria are known to be very sensitive to temperature changes; a temperature fluctuation of +/- 2^{o}C can result in a 30% reduction in biogas production (Zupancic and Grilc, 2012). To limit these effects, the temperature must be kept as constant as possible. For mesophilic digestion, a variation of up to +/- 3^{o}C is permitted (Zupancic and Grilc, 2012;

Shah *et al.*, 2017). Microorganisms are resilient and can operate outside of their optimal ranges, but at an extremely slowed rate which results in low biogas yields. For instance, thermophilic bacteria are active at temperatures as low as 45^{o}C and mesophilic bacteria can grow at temperatures as high as 47^{o}C (Zupancic and Grilc, 2012). At these overlaps, the respective methanogens are more or less inhibited., which needs to be avoided during actual operation.

<u>ii) Carbon to Nitrogen (C:N) Ratio</u>

Carbon to Nitrogen ratio is a useful property that provides an indication of microbial activity inside a reactor. Ideally, the C:N ratio is in the range of 20-30:1 (Verma, 2002; Zupancic and Grilc, 2012). A C:N ratio greater than 30 is indicative of high consumption of nitrogen by methanogenic bacteria (Jingura and Kamusoko, 2017b). Since nitrogen is used for protein synthesis in microorganisms, decreased availability of nitrogen results in reduced protein production. This lowers the growth rate of microorganisms, consequently slowing the rate of metabolism of the material (Zupancic and Grilc, 2012). Conversely, a low C:N ratio, possibly caused by a nitrogen-rich feedstock, may result in ammonia accumulation inside the reactor, likely causing inhibition (Verma, 2002). Ammonia tends to be toxic to mesophilic methanogens at concentrations over 3000 mg^{-1} and pH greater than 7.4 (Zupancic and Grilc, 2012). pH levels over 7.4 increases the toxicity of ammonia. Thermophilic methanogens are more sensitive to ammonia; as inhibition can occur at concentrations as low as 2200 mgL^{-1} (Zupancic and Grilc, 2012).

Ammonia accumulation can also be caused by the recirculation of leachate (or liquid from the digested material) used to dilute the feedstock before digestion (Zupancic and Grilc, 2012). This accumulation can be prevented by using a mixed feed, often referred to as co-digestion. This involves mixing a nitrogen-rich organic solid waste with sewage or animal manure (Verma, 2002; Sawyerr, Trois and Workneh, 2019).

<u>iii) pH</u>

Sub-optimal pH levels have an extremely detrimental effect on the process of anaerobic digestion. A poor pH level can result in an ultimate system failure (Verma, 2002). Methanogens are very sensitive to pH, and acidogens can be inhibited by acid accumulation (low pH) (Verma, 2002). These microbes require different pH levels to function optimally. Optimum pH for methanogens have been reported to be in a range of 6.5 to 8.5 whilst acidogens a pH range of 5.2 to 6.3 is documented (Verma, 2002; Weiland, 2010; Zupancic and Grilc, 2012). (Table 2.4-2). Acetogenesis can result in accumulation of organic acids inside a batch reactor due to shorter retention times of the phase. This can lower the pH inside the digester to below 5, subsequently poisoning the methanogens (Verma, 2002).

Addition of lime or recycled filtrate can be used to regulate the pH level inside the system (Verma, 2002).

Table 2.4 - 2: Parameters defining optimal AD environment (Zupancic and Grilc, 2012)

Parameter	Hydrolysis/ Acidogenesis	Methanogenesis
Temperature	25-35°C	Mesophilic: 30-40°C Thermophilic: 50-60°C
pH Value	5.2-6.3	6.7-7.5
C:N ratio	10-45	20-30
Redox potential	+400 to -300 mV	Less than -250 mV
C:N:P:S ratio	500:15:5:3	600:15:5:3
Trace elements	No special requirements	Essential: Ni, Co, Mo, Se

<u>iv) Retention time</u>

Retention time is the average time it takes for complete digestion of a certain amount of feedstock. It varies depending on the system technology used, operating temperature (mesophilic or thermophilic) and waste composition (Verma, 2002). The retention time for mesophilic digestion ranges from 10 to 40 days, whilst a thermophilic system has a retention time of less than 14 days (Verma, 2002).

<u>v) Mixing</u>

Mixing inside the reactor is necessary to ensure maximized substrate-microbial interactions in the system, as well as to ensure that there are no temperature gradients. Furthermore, it improves the ability of microorganisms to reach nutrients. Excessive mixing, however, can disrupt microbial activity, hence slow, carefully constructed mixing is preferred (Verma, 2002). The mixing equipment and speed of the mixer used are dependent on the type of reactor and the solids content inside the reactor (Verma, 2002).

<u>vi) Total solids content</u>

The total solids content of the AD process dictates the handling of waste, retention times, as well as the type of reactor used. Total solids content inside the reactor can be divided into three categories, namely; low solid (LS) (<10%), medium solid (MS) (15-20%) and high solid (HS) (22-40%) systems (Monnet, 2003; Yi *et al.*, 2014). Solids content is discussed further in the section describing types of digesters.

vii) Organic Loading Rate (OLR)/Volatile Solids (VS)

Organic loading rate (OLR) measures the biological conversion capacity of the anaerobic digestion system (Verma, 2002). It is measured as a kilogram of chemical oxygen demand (COD) or volatile solids (VS) per cubic meter of the reactor. Volatile solids (VS) is a measure of the organic matter in a sample, measured as solid content minus ash content after complete combustion of the material. Feeding a system above its sustainable OLR may result in reduced methane yield due to accumulation of inhibiting substances such as fatty acids in the digester (Verma, 2002).

viii) Feedstock

Feed composition is one of the most important determinants of biogas yield. Factors such as, quantity of biogas (particularly methane yield) that can be recovered from a particular feedstock, availability and cost of feedstock and how the material is degraded, are investigated. Table 2.4-3 shows the various feedstock and the percentage of methane yield that can be obtained from anaerobic digestion. Materials such as beet leaves can yield up to 84.8% methane in biogas. Food waste has a very high biodegradability and contains a high moisture content (Yi *et al.*, 2014). Various feedstock emanating from agricultural, industrial and community practices are reported in literature. (Table 2.4-4). The most abundant feedstock for anaerobic digestion is food waste, and it is derived from each of the above-mentioned waste generating practices. Often feedstock contains or may lead to accumulation of inhibitory substances. Common inhibitory substances include ammonia, hydrogen sulphide and heavy metals (Jingura and Kamusoko, 2017b).

ix) Anaerobic Co-digestion of feedstock

The conventional mono-feed in anaerobic digestion is not the most optimal solution for a stable, methane rich process (Chen and Neibling, 2014). There are several limitations to working with a single substrate such as imbalanced microbial population, volatile solids-moisture content, pH and C:N ratio which can lead to inhibition and process failure. The solution to this is in the form of a co-digestion feedstock. Substrate co-digestion is far superior and effective in terms of reactor performance, yield, solid reduction and nutrient balance (Rabii *et al.*, 2019). The combination of various substrates with complimenting physical and chemical characteristics have proven to increase the effectivity of the process (Sawyer, Trois and Workneh, 2019). The most common is the use of some form of manure (Table 2.4-3; Table 2.4-4) or sludge in co-digestion with one or more substrates. Manure is rich in nutrients and can act as an inoculum to the mixture.

Table 2.4 - 3: Typical methane composition for various feedstock (Anukam *et al.*, 2019)

Feedstock	CH_4 Composition (%)
Cattle manure	50–60
Pig manure	60
Poultry waste	68
Sheep dung	65
Horse dung	66
Grass	84
Wheat straw	78.5
Dried leaves	58
Barley straw	77
Beet leaves	84.8
Corn silage	54.5

Table 2.4 - 4: Potential feedstock for Anaerobic Digestion in South Africa (Smith *et al.*, 2011; Sawyerr *et al.*, 2019)

Sources	Various Feedstock
Agriculture	• Manure • Energy Crops • Algal Biomass • Harvest remains
Industry	• Food/beverage processing • Dairy • Starch industry • Sugar industry • Pharmaceutical industry • Cosmetic industry • Biochemical industry • Pulp and paper • Slaughterhouse/rendering plant
Communities	• OFMSW • MSW • sewage sludge • grass clippings/garden waste • food remains

2.4.6 Biomethane Potential (BMP) of Organic Feedstock

BMP tests are performed to determine the biodegradability of a biomass feedstock and their potential to produce methane (Sell *et al.*, 2010). This provides a baseline for the performance of an AD process and allows for optimization of the process (Jingura and Kamusoko, 2017b).

All biomass containing carbohydrates, proteins, fats, cellulose and hemicellulose as main components, can be used as feedstock as they are digestible (Weiland, 2010). Various feedstock commonly used in anaerobic digestion include Municipal Solid Waste (MSW), food waste, sewage sludge, animal manure, and energy crops (Jingura and Kamusoko, 2017b). Woody biomass despite their biodegradability are impractical for use without pre-treatment due to several factors, such as; low moisture content, degree of association between lignin and carbohydrates, particle size and cellulose crystallinity (Gumisiriza *et al.*, 2017; Sawyerr, Trois and Workneh, 2019). Pre-treatment assists with reducing lignin content and crystallinity of contents, making it possible for hydrolysis and absorption by microorganisms (Gumisiriza *et al.*, 2017).

Methods for Estimating Biomethane Yield

There are various methods that researchers use to estimate BMP. These methods can either be experimental or theoretical (Jingura and Kamusoko, 2017b). They follow the same principle, with some variation in the technical approach and experimental steps (Rodriguez, 2011). Choice of method therefore relies on associated cost and processing time (Jingura and Kamusoko, 2017b). Examples of experimental methods are the BMP test and spectroscopy, and elemental composition. Theoretical approaches include chemical composition and chemical oxygen demand (Jingura and Kamusoko, 2017b).

There is no standard procedure for undertaking BMP tests, which has resulted in various commonly used methods which differ in operational conditions being reported in literature (Jingura and Kamusoko, 2017b). Such methods include the German standard procedure, Verein Deutscher Inenieure (VDI method), the Hansen method and the Moller method (Pham *et al.*, 2013). The Moller method operates under mesophilic conditions, the Hansen method under thermophilic conditions, and the VDI method works under both conditions. There are two types of BMP tests: the conventional BMP test and the automatic BMP test (Jingura and Kamusoko, 2017b).

Both the conventional and the automated tests utilize the same principle to determine the volume of methane produced (Jingura and Kamusoko, 2017b). The organic feedstock is mixed with an inoculum under specific operating conditions and the gas produced is physically quantified by manometric or volumetric methods (Jingura and Kamusoko, 2017b). The conventional BMP test is, however, time consuming and resource intensive but it is a much simpler and cheaper approach with great repeatability (Rodriguez, 2011). The automatic BMP test uses less labour and is cost effective in the long run (Jingura and Kamusoko, 2017b).

Spectroscopy tests determine the absorbance, transmission or fluorescence of radiation in the visible (VIS), ultraviolet (UV) and infrared (IR) range (Jingura and Kamusoko, 2017b). A basic spectroscopic setup includes a “radiation source, a wavelength selector, sample cell, reagent dosing unit (for VIS spectrometry), detector, as well as data treatment and read out unit" (Jingura and Kamusoko, 2017b). One setback with this approach is the high associated cost.

The theoretical approach is most useful where lab access is limited. Regression models are used to estimate the potential methane yield. This is based on the assumption of total degradation of the organic matter and consequently the yield obtained from this method is usually over estimated (Jingura and Kamusoko, 2017b).

<u>Methane potential estimation from elemental compositional analysis</u>

Elemental composition can be used to calculate methane potential of the substrates. Buswell's equation illustrated below is used to estimate methane potential from ultimate analysis. This equation works on the assumption that all the organic material is converted to biogas (Buswell and Neave, 1930). Equation 2.3-6 gives the biogas potential, and the subsequent Equation 2.3-7 calculates the total methane generated in (*l* CH_4/g VS at STP).

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

Equation 2.3.2-1

$$B_{th}(\frac{m^3}{kgVS}) = \frac{22.415a}{12a + b + 16c + 14d}$$

Equation 2.3.2-2

$$M_{th}(\frac{m^3}{kgVS}) = \frac{22.415(\frac{4a + b - 2c - 3d}{8})}{12a + b + 16c + 14d}$$

Equation 2.3-3

The general formula is in the form $C_aH_bO_cN_dS_e$. The dimensionless coefficients a, b, c, d are determined using the equations listed below.

$$a = \frac{(\frac{\%C}{Molar\,Mass\,C})}{L},\ b = \frac{(\frac{\%H}{Molar\,Mass\,H})}{L},\ c = \frac{(\frac{\%O}{Molar\,Mass\,O})}{L},\ d = \frac{(\frac{\%N}{Molar\,Mass\,N})}{L}$$

Equation 2.3-4

The constituents %C, %H, %O and %N represents the proportion of C, H, O, N in the substrates. The variable L is calculated as shown in Equation 2.3.6-5 below. M represents the element with the least number of moles in a sample which is normally N, such that the coefficient d is equal to 1.

$$L = \frac{\%M}{\text{Molar Mass M}}$$

Equation 2.3-5

For co-digestion applications, for substates V and W, fed into a reactor in a proportion I% of V and J% of W, where I and J adds to 100%, the following equations apply. The first step is the recalculation of the elemental compositions C, H, N, O to represent the proportion of the substrates in the feed. Equation 2.3.6-6 shows the calculation for Carbon (C), the rest of the elements are calculated in a similar manner.

$$\%C_{(I\%V+J\%W)} = \%Ia_V \frac{\text{Molar Mass C}}{\text{Molar Mass V}} + \%Ja_W \frac{\text{Molar Mass C}}{\text{Molar Mass W}}$$

Equation 2.3-6

After recalculating the elemental composition of the substrates, the coefficients a, b, c, d are calculated in a similar manner to Equation 2.3-8. A calculation for coefficient a is shown below.

$$a_{(I\%V+J\%W)} = \frac{\dfrac{\%C_{(I\%V+J\%W)}}{\text{Molar Mass C}}}{L_{(I\%V+J\%W)}}$$

Equation 2.3-7

Where L is calculated as

$$L_{(I\%V+J\%W)} = \frac{\%M_{(I\%V+J\%W)}}{\text{Molar Mass M}_{(I\%V+J\%W)}}$$

Equation 2.3-8

<u>Biomethane potential estimation using nutrient compositional analysis</u>

The equation described below estimates methane potential from nutritional analysis for a substrate assumed to contain only carbohydrates, lipids and proteins. This equation was proposed by Angelidaki and Sanders (2004) and it is noted below for single substrates.

$$CH_4\,\text{Yield}\,\left(\frac{m^3}{\text{kg VS}}\right) = 0.496X' + 1.014Y + 0.415Z$$

Equation 2.3-9

Where:

X’ = fraction of proteins in substrate VS

Y = fraction of lipids in substrate VS

Z = fraction of carbohydrates in substrate VS

For co-digestion applications, for substrates V and W, with compositional contribution of I% of V and J% of W, Equation 2.3.6-9 is adapted as follows

$$CH_4\ \mathrm{Yield}\left(\frac{m3}{kg\ VS}\right) = 0.496(\%I\times\%prot_V + \%J\times prot_W) + 1.014(\%I\times\%lip_V + \%J\times lip_W) + 0.415(\%I\times\%carb_V + \%J\times carb_W)$$

Equation 2.4-10

Where:

Prot = proportion of proteins

Carb = proportion of carbohydrates

Lip = proportion of lipids

Biomethane Yield (South African context)

South Africa has vast feedstock resource available for the production of methane using anaerobic digestion (Table 2.4-5). Typical feedstock found in South Africa include animal manure, Organic Fraction of MSW, food waste and sewage sludge (Sawyerr *et al.*, 2019). A large amount of this waste is highly accessible for beneficiation. Animal manure can be obtained from cattle or livestock farms, food waste from eating areas such as restaurants or large malls and sewage sludge can be obtained from sewage treatment facilities (Sawyerr, Trois and Workneh, 2019). Fruit and Vegetable waste can be obtained from the major fruit and vegetable market outlets available to the province at a cost of transportation only. Regarding energy crops, they are usually grown for this purpose and the important qualifying parameters are harvest time and methane yield per hectare (Weiland, 2010). Another feedstock not yet available to, but suitable for South Africa is cassava plant (Sawyerr, Trois and Workneh, 2019). Cassava plant, its peel and by products when co-digested with other feedstock produced high methane yields, making it a favourable energy plant that can be used in various communities (Sawyerr, Trois and Workneh, 2019). The author indicated that this crop is capable of growing in South Africa and thriving under current environmental conditions. There are also a wider variety of biomass feedstock that can be used to generate a large amount of methane (Table 2.4-6). Most of the feedstock compiled in Table 2.4-6 is available in South Africa and accessible for anaerobic digestion.

Biogas companies currently operating in the country include Bio2Watt (Pty) Ltd, Mpfuneko Community Support (MCS) and Netherlands Wild Goose Dutch Development Organisation, Biogas SA and BiogasPro (Kigozi, Aboyade and Muzenda, 2014)

Table 2.4 - 5: Biomethane yield for available South African Feedstock

Feedstock	Production (South Africa) (tons/year)	Biomethane yield m^3/kg VS	Method	Reference
Agriculture				
Cattle manure	136 161	0.575	Theoretical	(Sawyerr, Trois and Workneh, 2019)
		0.115	BMP	(Tawona, 2015)
Pig manure		0.3 - 0.5	BMP	(Weiland, 2010; Wilkinson, 2011)
Chicken manure		0.35 - 0.6	BMP	(Wilkinson, 2011)
Maize		0.399	BMP	(Jingura and Kamusoko, 2017b)
Community				
Banana peels	371 385	0.274 – 0.322	BMP	(Gunaseelan, 2007)
Sewage sludge		0.3 - 0.303	BMP	(Wilkinson, 2011; Arhoun *et al.*, 2019)
Grass		0.123		(Sawyerr *et al.*, 2019)
Branches		0.140		(Sawyerr *et al.*, 2019)
Pineapple peels		0.357	BMP	(Weiland, 2010)
Human Excreta		0.02 – 0.028	BMP	(Andriani *et al.*, 2015)
MSW		0.31 – 0.4		(Kigozi, Aboyade and Muzenda, 2014)
FV	1387	0.403 - 0.533	BMP	(Arhoun *et al.*, 2019; Sawyerr *et al.*, 2019)
Co-digestion				
CM:FV	80:20	0.526	Theoretical	(Sawyerr, Trois and Workneh, 2019)
		0.146	BMP	(Tawona, 2015)
CM:VW	75:25	0.437	BMP	(Sandhu and Kaushal, 2019)
CT:FV	20:80	0.526	Theoretical	(Sawyerr, Trois

				and Workneh, 2019)
FV:MSS	40:60	0.455	BMP	(Arhoun *et al.*, 2019)

CM: Cattle manure; CT: Cassava tubes; FV: Fruit and vegetables; MSS: Municipal Sewage sludge; MSW: Municipal solid waste; VW: Vegetable waste

Table 2.4 - 6: Methane yield for various biomass feedstock (Sawyerr et al., 2019)

Biomass	Methane yield (m^3/kg VS)	Reference
OMSW		
HS-OMSW	0.390	(Cecci, Traverso and Cescon, 1986)
SC-OMSW	0.403	(Mata-Alvarez *et al.,* 1990)
SS-OMSW	0.399	(Sawyerr et al., 2019)
Fruit and vegetable solid waste and leaf		
Potato waste	0.426	(Stewart, Bogue and Barger, 1984)
Carrot waste	0.417	(Shen *et al*., 2013)
Banana fruit and stem	0.529	(Murphy *et al.,* 2011)
Tomato processing waste	0.420	(Sarada and Joseph, 1994)
Banana peeling	0.409+/-0.002	(Izumi *et al*., 2010)
Grassy Biomass		
Sorghum	0.420	(Gunaseelan, 2004)
Corn stover	0.360	(Gunaseelan, 2007)
Paddy straw	0.367	(Mshandete *et al*., 2006)
Milet straw	0.390	(Mahamat *et al*., 1989)
Wheat straw	0.383	(Hashimoto, 1986)
Woody biomass		
Iponnoea stem	0.426	(Seppälä, Paavola and Rintala, 2007)
Poplar wood	0.330	(Gunaseelan, 2004)
Pre-treated vine shoot	0.315	(Odlare, 2005)
Weed biomass		
Lantaria treated with NaOH + cow manure	0.236	(Dar and Tandon, 1987)
Partially decomposed Ageratum	0.241	(Kanwar and Guleri, 1995)
Parthenium treated with NaOH	0.236	(Sawyerr et al., 2019)
Marine biomass		
Ulea and *Chaeromarpha*	0.480	(Hansson, 1981)
Ulea	0.330	(Bohutskyi and Bouwer, 2013)
Maerocystis pyrifera	0.310	
Freshwater Biomass		
Pisitia	0.410	(Nipaney and Panholzer, 1987)
Water hyacinth treated with NaOH	0.362	(Chynoweth *et al.*, 1982)

*Citations not included in the reference list since they were not compiled in this work

2.4.6 Types of Digesters

There are several ways to distinguish between the different types of digesters. In this report the classification criterion is as follows.

- Single-stage systems (SS)
- Multi-stage systems (MS)
- Batch systems

These are further categorized as either wet or dry depending on the total solid content of the slurry in the reactor. Therefore, the classification is further denoted as SSLS for single stage low solids, SSHS for single stage high solids, multi-stage low solids (MSLS) and multi-stage high solids (MSHS). High solids digesters are generally more expensive than single-stage due to the elevated costs of pumping and piping equipment, as well as maintenance (Verma, 2002). Whereas operating a low solids reactor normally involves a large volume of water, which increases the overall volume of the reactor (Verma, 2002). This results in an increased cost of dewatering after digestion.

i) Single Stage Low Solids system (SSLS)

One stage, wet complete mix systems as its often referred to in literature, is the most commonly used type of reactor, with usage dating back decades where it was used mainly for the stabilization of bio-solids from wastewater treatment facilities (Vandevivere, De Baere and Verstraete, 2016). This system uses a slurry feed of 3.2-15% total solids (TS) diluted with water and digested in a continuously stirred tank reactor (CSTR) (Vandevivere, De Baere and Verstraete, 2016). This reactor configuration is commonly used for moisture rich municipal organic waste with high volatile fatty acids which often leads to inhibition (Markphan *et al.*, 2020). Complete mixing is conducted in a large cylindrical chamber (tank) which is connected to a mixing device and can be configured to operate under mesophilic and thermophilic conditions (Chen and Neibling, 2014). One of the earliest full-scale plants using this technology is the Waasa plant in Finland, built in 1989 (Figure 2.4-3).

The system is equipped with a pulper where the waste is shredded, homogenized and diluted in sequential batches, water is recycled and mixed together with the fresh feedstock to achieve a total of 10-15% TS in the feed stream (Vandevivere, De Baere and Verstraete, 2016). The digester is also equipped with vertical impellors to ensure complete mixing inside the reactor and prevent sinking of solid particles. The sinking of material to the bottom of the reactor is a common problem associated with this type of reactor. Accumulation of material at the bottom can damage impellors and if collected in the streams can damage pumps, hence the pre-treatment step needs to be vigorous and often complex. The solution to this problem involves removing large material which results in loss of 15-25% biodegradable material from mechanically sorted Organic Fraction of Municipal Solid Waste (OFMSW), resulting in reduced biogas yield (Vandevivere, De Baere and Verstraete, 2016). A floating "scum layer" also accumulates at the top, creating three layers inside the reactor, which can be several meters thick and limits effective mixing (Monnet, 2003; Vandevivere et al., 2016).

The advantages of using a CSTR (SSLS) include good mixing inside the digester, good dilution of inhibitors and affordability and the shortfalls include among others a large water usage and short circuiting of material depending on equipment design and material digested (Vandevivere, De Baere and Verstraete, 2016). Other examples of SSLS systems which are commonly used in farms are covered lagoon digesters, low solids plug flow digesters, and fixed-film digesters (Chen and Neibling, 2014).

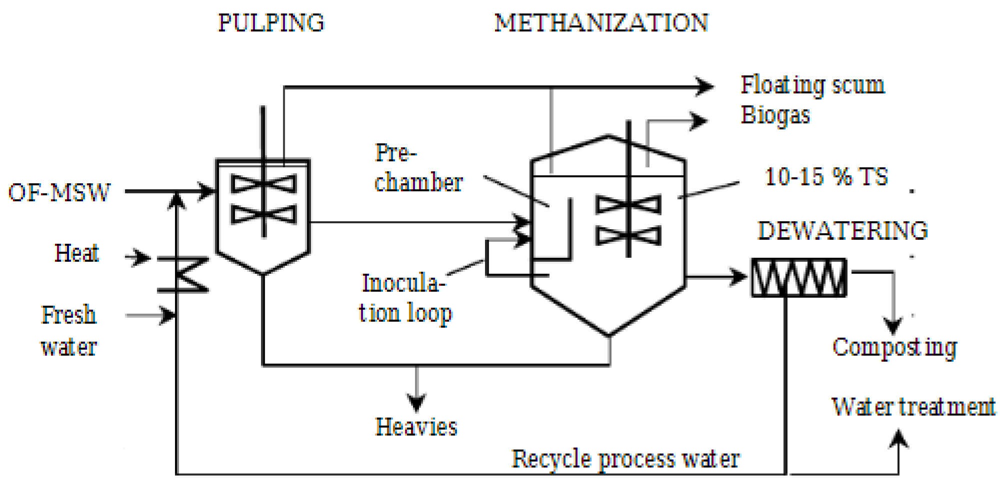


Figure 2.4 - 3: Typical design of a SSLS system (Vandevivere, De Baere and Verstraete, 2016)

<u>ii) Single Stage High Solids system (SSHS)</u>

The SSHS system allows a total solids feed in the range 20-40% (Vandevivere, De Baere and Verstraete, 2016). This system generally has higher biogas yields than single stage low solids systems. These systems have been used extensively in France and Germany for digestion of mechanically sorted OFMSW (Vandevivere, De Baere and Verstraete, 2016). The pre-treatment process for SSHS is not as stringent as for SSLS, as it involves screening of particles over 40 mm in size which can be achieved through drum screening (common for mechanically sorted OFMSW) or via shredders for source-separated bio-waste (Vandevivere, De Baere and Verstraete, 2016). Unlike SSLS, a plug flow arrangement is used (Figure 2.4-4). Since the slurry is highly viscous, a plug flow arrangement allows for better treatment of waste with no moving parts inside the reactor that can get damaged during the process. Setbacks arise only if the fresh waste mixes with already fermenting mass, affecting the inoculation process, and causing local overloading and acidification. The important advantage for this system is the retention of volatile solids which makes it more efficient than the CSTR and the advantage of a smaller reactor size. Table 2.4-7. However, dealing with solids rich slurry is very difficult and requires sophisticated and expensive machinery to operate.

Figure 2.4-4 shows the three feasible plug flow designs (Dranco, Kompogas and BRV, and Valorga designs) for SSHS systems used at an industrial scale. In the Dranco design, mixing is achieved by extracting a recirculating waste stream at the bottom of the reactor, mixing it with fresh feed, and then pumping it to the top. One part of freshwater is mixed with six parts of digested waste. This technology works best for waste feed with 20-50% TS (Vandevivere, De Baere and Verstraete, 2016).

Kompogas and BRV designs work in a similar manner, but in a horizontal position. In this design, slowly rotating impellors inside the cylindrical reactors facilitate the flow and ensure homogenization, degassing and re-suspending of heavy components (Vandevivere, De Baere and Verstraete, 2016). This reactor design works well at around 23% TS.

The Valorga system is different such that flow inside the vertical reactor is horizontal rather than along the reactor length. Mixing inside this reactor is achieved via the introduction of biogas at high pressure at the bottom of the tank every 15 minutes through a network of injectors. Process water is recirculated to achieve a solid content of 30% TS (Vandevivere, De Baere and Verstraete, 2016). A technical drawback of this system is that biogas injection ports at the bottom of the reactor tend to clog with sedimentation of waste which is common for fine wastes with <20% TS (Vandevivere, De Baere and Verstraete, 2016).

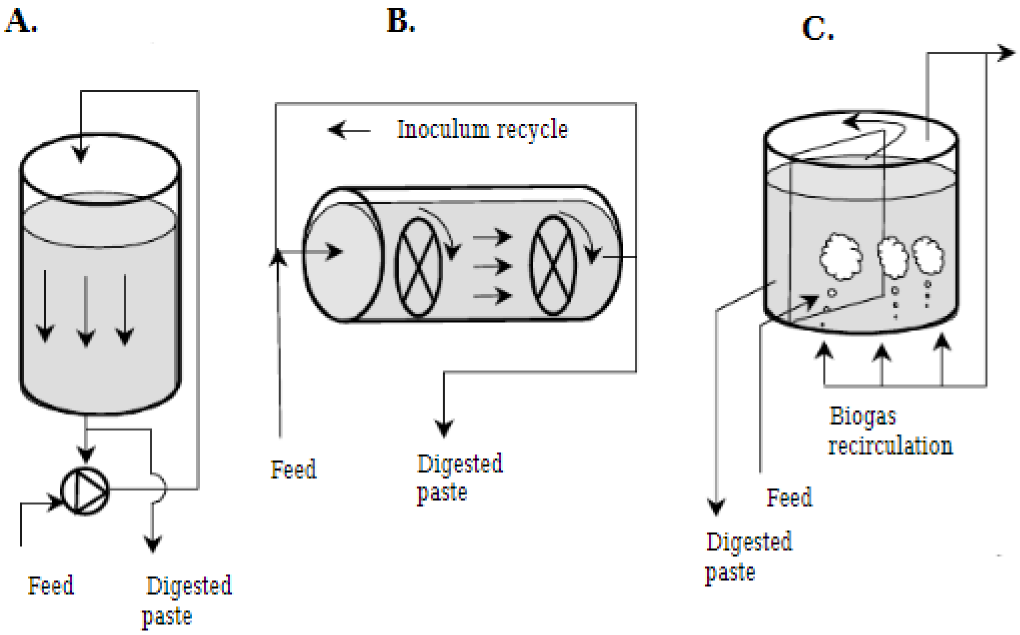


Figure 2.4 - 4: Illustrations of the different low solids digester designs (A- Dranco design, B- Kompogas and BRV designs, C- Valorga design) (Vandevivere, De Baere and Verstraete, 2016)

Table 2.4 - 7: Advantages and disadvantages of SSHS systems (Vandevivere, De Baere and Verstraete, 2016)

Criteria	Advantages	Disadvantages
- Technical :	- No moving parts inside reactor - Robust (inerts and plastics need not be removed) - No short-circuiting	- Wet wastes (< 20 % TS) cannot be treated alone
- Biological :	- Less VS loss in pre- treatment - Larger OLR (high biomass) - Limited dispersion of transient peak concentrations of inhibitors	- Little possibility to dilute inhibitors with fresh water
- Economical & Environ-mental :	- Cheaper pre-treatment and smaller reactors - Complete hygienization - Very small water usage - Smaller heat requirement	- More robust and expensive waste handling equipment (compensated by smaller and simpler reactor)

iii) Multi-stage systems

The rationale for multi-stage systems is that the overall conversion of waste occurs from a sequence of biochemical reactions which do not share the same environmental conditions. Therefore, separating these stages allow for optimization of process parameters for hydrolysis-acidogenesis and methanogenic microorganisms which then leads to increased reaction rate and methane yield (Rabii et al., 2019). The result is that for very acidic substrates which often lead to accumulation of acid after acidogenesis does not lead to inhibition of methanogens inside the reactor. The range of operating conditions for the various microorganisms is stated in section 2.3.5.

This type of system is not used extensively, and accounts only for 10% of reactors used in industry (Vandevivere, De Baere and Verstraete, 2016). For a two-stage system, hydrolysis-acidification takes place in the first reactor and acetogenesis-methanogenesis in the second reactor (Vandevivere, De Baere and Verstraete, 2016). An example of such a system is the Schwarting-Uhde process. (Figure 2.4-5). Researchers such as Monnet (2003) stated a preference for the first three processes to be contained in the first reactor and methanogenesis to occur in the second reactor. The rate in the first reactor is limited by hydrolysis, whilst the second is limited by methanogenic microbial growth. It is because of this, oxygen is often introduced in the first reactor to speed up the hydrolysis process, and a specifically designed reactor with improved biomass retention is reserved for methanogenesis (Vandevivere, De Baere and Verstraete, 2016). Biomass retention can be used to determine the biological reliability of the reactor (Vandevivere, De Baere and Verstraete, 2016).

Some of the advantages of a multi-stage configuration is the increase in process stability and yield, reduction in COD and a high organic loading rate (Rabii et al., 2019; Markphan et al., 2020). Two-stage systems with biomass retention excel better at handling waste with high amounts of inhibitors and N_2, whereas single-stage systems tend to be unstable under such conditions (Vandevivere, De Baere and Verstraete, 2016). Despite the aforementioned theorised advantages, a two-stage process has not always provided better results than a single-stage system (Vandevivere, De Baere and Verstraete, 2016). This explains why a single-stage system is the preferred choice in industries. Some disadvantages of the technology include high construction and maintenance cost and they tend to be very complex (Vandevivere et al., 2016; Rabii et al., 2019; Markphan et al., 2020).

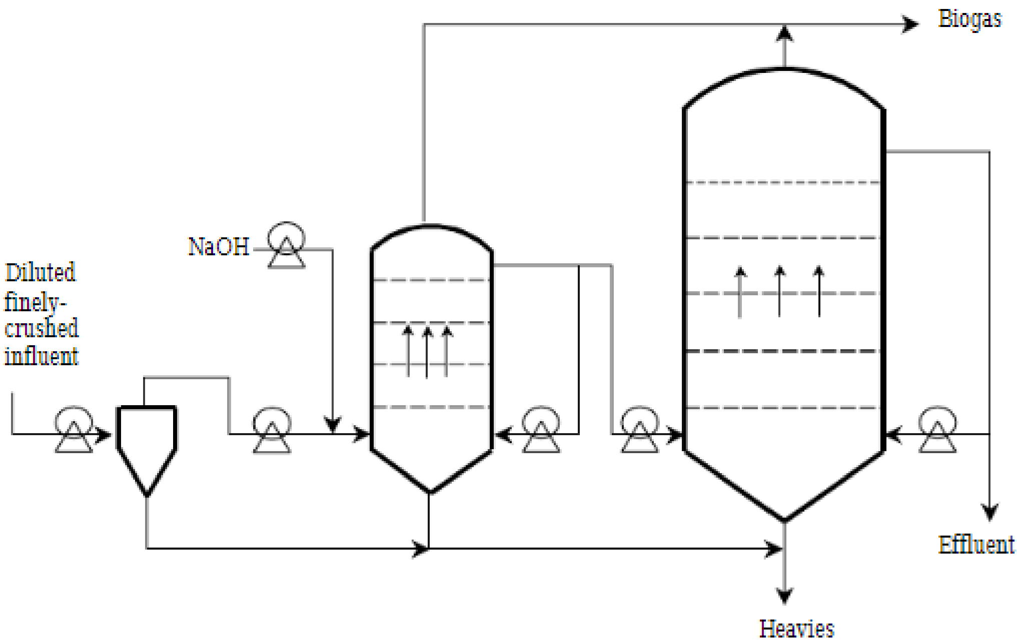


Figure 2.4 - 5: Process flow diagram of a Schwarting-Uhde process (Vandevivere, De Baere and Verstraete, 2016)

iv) Batch systems

A batch system is comparable to a landfill in a box, but with a 50 to 100 fold higher biogas production rate (Vandevivere, De Baere and Verstraete, 2016). Unlike landfills, batch systems are run at high temperatures, and leachate is recirculated, which allows dispersion of inoculant, nutrients and acids (Vandevivere, De Baere and Verstraete, 2016). In a batch system, fresh waste at a slurry feed of 30-40% is fed once and allowed to go through all the degradation steps in dry mode (Monnet, 2003). The retention time for a batch system is 30 to 180 days (Chowdhury and Fulford, 1992). This type of system is least used in industry, but their ability to handle coarse and heavy contaminants with a low investment cost, simple design and process control make them a good fit for developing countries (Vandevivere, De Baere and Verstraete, 2016). There are three types of batch systems; single stage system, sequential system and up-flow anaerobic sludge blanket system (UASB) (Figure 2.4-6).

In a single-stage design, leachate is recirculated to the top of the reactor from which it is produced (Vandevivere, De Baere and Verstraete, 2016). The leachate is collected under the reactor and then sprayed on the top surface of the waste. A limitation of this system is the perforated floor which becomes clogged with material, blocking access to the leachate, and this is solved by limiting the thickness of the fermenting waste to four meters and mixing feed with bulking material (Vandevivere, De Baere and Verstraete, 2016).

In a sequential batch system, the leachate from the first reactor, which is rich in organic acids, is recirculated into the next reactor where methanogenesis takes place (Vandevivere, De Baere and Verstraete, 2016). The leachate of the second reactor is sent back to the first reactor after recovering the acids and adding pH buffering bicarbonates (Vandevivere, De Baere and Verstraete, 2016). In the hybrid batch-UASB design, the second reactor where methanogenesis takes place is replaced with an up-flow anaerobic sludge blanket (UASB) reactor (Vandevivere, De Baere and Verstraete, 2016). This reactor is suitable for treating liquid effluents with high levels of organic acids at high loading rates (Vandevivere, De Baere and Verstraete, 2016). The advantages of a batch system include low technology, it can handle high solids and it is not complicated. The disadvantages include a low biogas yield, longer retention time and clogging (Chowdhury and Fulford, 1992).

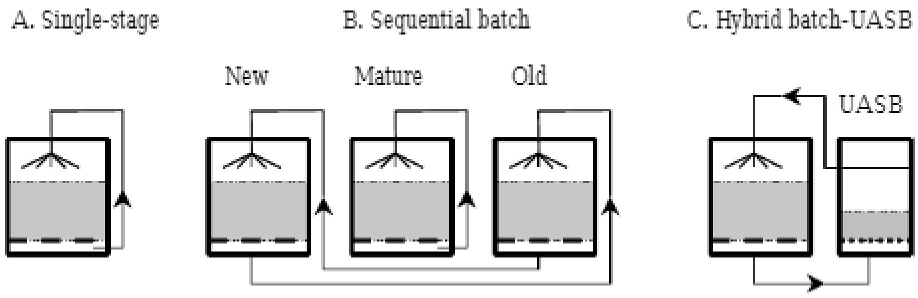


Figure 2.4 - 6: Configuration of leachate recirculation patterns in different batch systems (Vandevivere, De Baere and Verstraete, 2016)

2.5 Chapter Summary

This chapter presented available information pertaining to the waste management practices in South Africa and the world. A gap was identified in the lack of source separation of waste which makes it difficult to make use of the various waste management solutions available including Anaerobic Digestion. The chapter concluded an in-depth presentation of the Anaerobic Digestion technology. The next chapter presents a case study assessing the major landfills in eThekwini municipality in South Africa and providing a stepwise solution for various scenarios that are applicable to landfills through a simulation using the WROSE model. Mechanical Biological Treatment (MBT) solution, utilising AD as a biological component is assessed.

CHAPTER 3 - Assessment of Anaerobic Digestion for treatment of Waste in a South African Municipality – A case study

3.1 Introduction

This chapter provides an assessment into the waste disposal trends in eThekwini municipality's 4 major landfills and furthermore provides an analysis of the potential rehabilitation of the landfills through the utilisation of the AD technology as the biological component of a Mechanical Biological Treatment (MBT) facility. Various solution scenarios are presented using the Waste and Resource Optimization Scenario Evaluation model (WROSE) and discussed. The data presented in this chapter was obtained from Durban Solid Waste (DSW) and analysed on Microsoft Excel.

3.2 Landfilling trends in eThekwini municipality

The waste is handled by Durban Solid Waste department (DSW) from households via the transfer station shown in Figure 3.2-1 and sent to the various landfills in the area. The major landfills in eThekwini municipality are Bisasar landfill, Marianhill landfill, Buffelsdraai landfill and Illovu landfill. They are managed by Durban Solid Waste department. This information is very useful in establishing new and viable technologies needed to implement a mechanical and biological treatment facilities at local landfills. Figure 3.2-1 shows the state of waste from source prior to landfilling. This outlook is a result of the non-existence of measures such as source separation of waste, resulting in a mixed waste stream that is very difficult and costly to divert. The waste at this facility is facilitated by the excavators and sent into compaction trucks with minimum human interaction.

Figure 3.2 - 1: Waste at electron road transfer station in Durban

3.2.1 Analysis of landfilling data

A waste disposal analysis was done on the waste information provided by Durban Solid Waste department (DSW). The information contained recorded waste masses as it enters the various landfills in the area. The analysis was performed on the four major landfills in the area namely, Bisasar road landfill, Mariannhill landfill, Buffelsdraai landfill and Illovu landfill. Table 3.2-1 presents the disposal data for the various landfills investigated. Bisasar road landfill is the oldest and largest landfill in the area, commissioned in 1980, and it is followed by Mariannhill landfill and Buffelsdraai landfill (Table 3.2-2). Bisasar road is nearly depleted of space and has in recent years prohibited from receiving general/household waste which has since been diverted to other landfills. Since the reduction in landfilling at Bisasar, other landfills such as Buffelsdraai and Mariannhill landfills have observed an increase in disposal rates (Table 3.2-2). With over 3000 tonnes per day in 2010, the disposal rate for Bisasar road has dropped to just over 1000 tonnes per day. The younger landfills, Buffelsdraai and Illovu landfills has grown to over 700 and 581 tonnes per year, respectively in 2019. The estimated remaining landfill space for the two landfills is 52 and 32 years respectively.

Table 3.2 - 1: Average waste received (t/d) (Source: Courtesy of DSW)

Landfill	Waste received (t/d)		
	2019	2015	2010
Illovu	581.53	244.69	n/a
Marriannhill	1065.67	780.37	527.13
Buffelsdraai	706.53	331.28	254.67
Bisasar	1035.26	2119.43	3057.74

Table 3.2 - 2: Landfill specifications data (Source: Courtesy of DSW)

Landfill	Illovu	Marriannhill	Buffelsdraai	Bisasar
Design airspace availability (m^3)	9 660 000	4 400 000	43 026 691	25 000 000
Remaining Airspace (m^3)	8 786 615	102 500	40 185 392	330 000
Remaining landfilling years	31.7	0.9	52	0.9
Remaining design life	32	1	52	1
Rehabilitated areas (m^3)	85 700	193 000	232 350	360 000
Commission year	2014	1997	2006	1980

3.2.2 Waste Stream analysis

Analysis of the various waste types disposed of at the major landfills was conducted. The data was analysed for the years 2001, 2010, 2015 and 2019 to cover the major changes that occurred to the disposal trends at the major landfills over the 20-year period. Durban Solid Waste (DSW) waste stream accounted for the most landfilling from 2001 to 2015. DSW waste contains a variety of household waste which is collected weekly around the city. The waste is transferred to the landfill via the transfer station in the state displayed in Figure 3.2-2. Much of this waste is food waste coming from households. In 2019, sand and cover material had the largest disposal followed by DSW, builders rabble and general solid waste. Builders rabble and general solid waste constituted between 7-13% each in the years under sturdy. The rest of the waste stream which constitute recyclables, tyres and mixed loads did not record many changes in the past 20 years. In this sturdy, much emphasis is towards the organic waste which composes garden waste and the organic fraction of DSW waste. The diversion of this waste can benefit greatly the end life as well as emission potential of the various landfills.

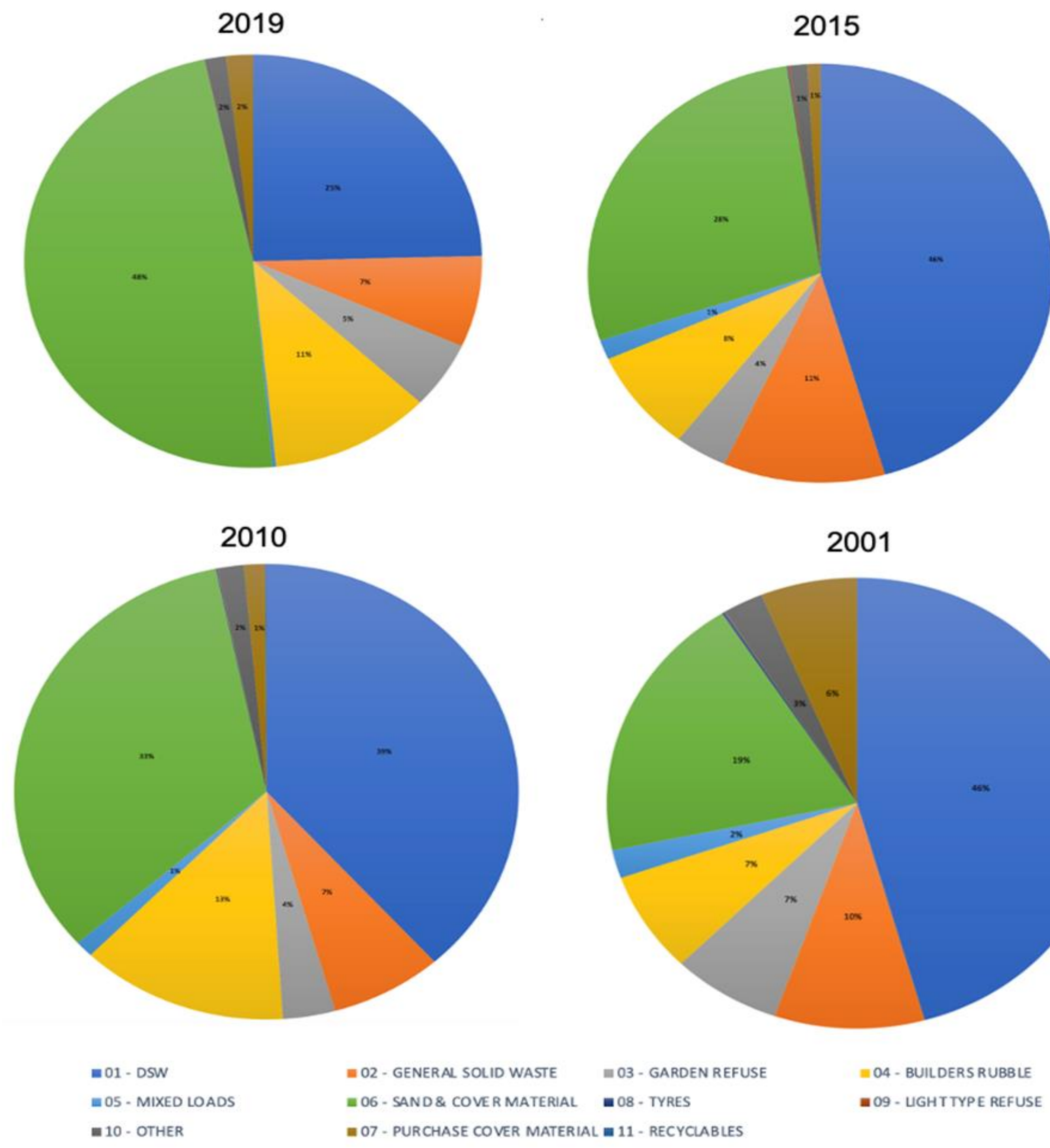


Figure 3.2 - 2: Waste stream analysis for the various landfills for the past 20 years (Mukwevho, 2021)

3.2.3 Analysis of organic waste at landfill

Organic waste includes garden waste and the organic fraction of Durban Solid Waste (DSW). DSW as already stated comprises of household and general waste which can be diverted. The analysis was conducted over a period of 5 years from 2015 to 2019. As discussed in the previous section, from 2015 most of the waste was diverted from Bisasar road leading to an increase in disposal rates in the other landfills. Much of the DSW waste is disposed of in Marriannhill and Buffelsdraai landfills between 2016 and 2019, Figure 3.2-3, whilst most of the garden waste was still landfilled in Bisasar road as well as Marriannhill and Illovu landfills (Figure 3.2-4). The DSW component is very high compared to garden waste with over 250 000 tonnes disposed per year at Mariannhill landfill and due to the large amount of food waste, this makes it a much greater pollutant and subsequently a better feedstock for biological treatment. Garden waste can disintegrate but due to the high lignocellulose, it does not decompose effectively. Diverting these waste components will not only improve the landfill space but also the sanitary state of the landfills.

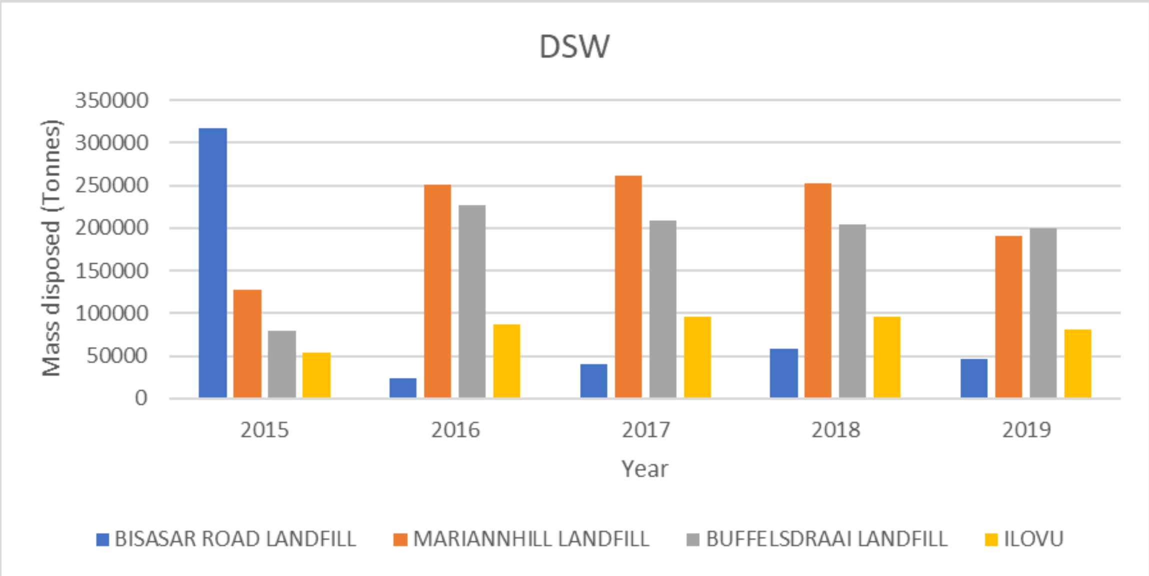


Figure 3.2 - 3: DSW waste disposal analysis

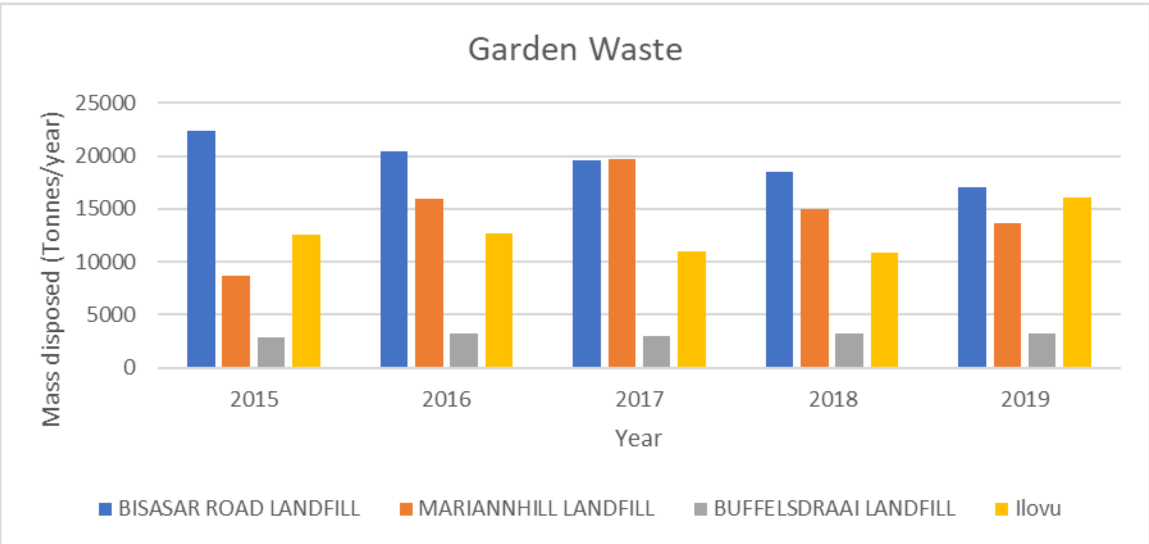


Figure 3.2 - 4: Garden waste disposal analysis

3.3 Waste and Resource Optimization Scenario Evaluation Model (WROSE)

In assessing the possible solutions and their outcomes for the treatment and valorisation of the OFMSW and F&V waste in eThekwini, the WROSE model is a useful tool for simulation purposes. This tool was developed at the University of KwaZulu-Natal by Prof. C Trois (Trois and Jagath, 2010) with the aim to assist various municiplities and the private sector to achieve zero waste goals. The model evaluates the possible outcomes for the various waste management practices proposed. The key outcomes from the model are:

- Waste stream analysis
- Quantification of Green House Gas (GHG) emissions reduction
- Scenario analysis
- Landfill diversion rates (Landfill space savings)
- Economic analysis (CAPEX, OPEX)

The model is Microsoft Excel based and uses South African data and emission factors which makes it applicable to developing countries (Kissoon, 2018). The WROSE model covers a wide range of waste management strategies available to developing countries such as landfilling, landfill gas extraction (landfill gas to energy technology), recycling, anaerobic digestion (AD) and composting. The above strategies to which the model was founded are broken down into the following scenarios, applicable to this research (Trois and Jagath, 2010 cited in Kisson, 2018)

Scenario 1: Landfilling of unsorted, untreated Municipal Solid Waste (MSW)

Scenario 2: Landfilling of unsorted, untreated MSW with landfill gas recovery

Scenario 3: Mechanical Pre-treatment of MSW with recycling and landfill gas recovery

Scenario 4: Mechanical Biological Treatment (MBT): Recycling, AD and landfill gas recovery

Scenario 5: Mechanical Biological Treatment (MBT): Recycling, composting and landfill gas recovery

3.3.1 Waste management scenarios explained

The following scenarios cover the waste management technologies available to developing countries and they cover landfills, landfill gas to energy, material recovery and anaerobic digestion. The following table summarises the technology (Table 3.3-1) and pictured in Figure 3.3-1.

Table 3.3 - 1: Waste management scenarios explained

SCENARIOS	Waste Management Strategy				
	Landfilling	LGR	Recycling	Anaerobic Digestion	Composting
1	X				
2		X			
3		X	X		
4		X	X	X	
5		X	X		X

Scenario 1

The first scenario is the conventional method followed by most municipalities as the cheapest and easy to use method. It is regarded as the baseline scenario. The waste is collected and without any treatment or processing sent for landfilling.

Scenario 2

Scenario 2 is the same as scenario 1 but with the exception of a much more sophisticated landfill gas to energy technology placed at the landfill to harness the biogas released by the biogenic waste anaerobically. This gas can be processed and relayed back to generate electricity into the grid. In South Africa, eThekwini municipality, Bisasar and Mariannhill landfills boasts of this technology.

Scenario 3

This scenario introduces the first Pre-Treatment technology applied prior to landfilling. The waste is separated at a Mechanical Recovery Facility (MRF), and the recylable portion is recovered whilst the rest of the waste including biogenic waste is sent for landfilling.

Scenario 4

This scenario assesses a closed loop waste management strategy. The first step is Pre-Treatment with recylable waste first recovered by the Mechanical Recovery Facility (MRF), from which the biogenic waste is diverted through Anaerobic Digestion (AD). Finally, the landfill is equipped with a landfill gas to energy technology. A possible scenario that involves AD and MBT as simulated by the WROSE model would look like scenario 4.

Scenario 5

This scenario is much similar to scenario 4 but with the exception to the biological treatment. Instead of diverting the biogenic waste to AD treatment, it is rather composted.The scenario also has landfill gas recovery and recycling.

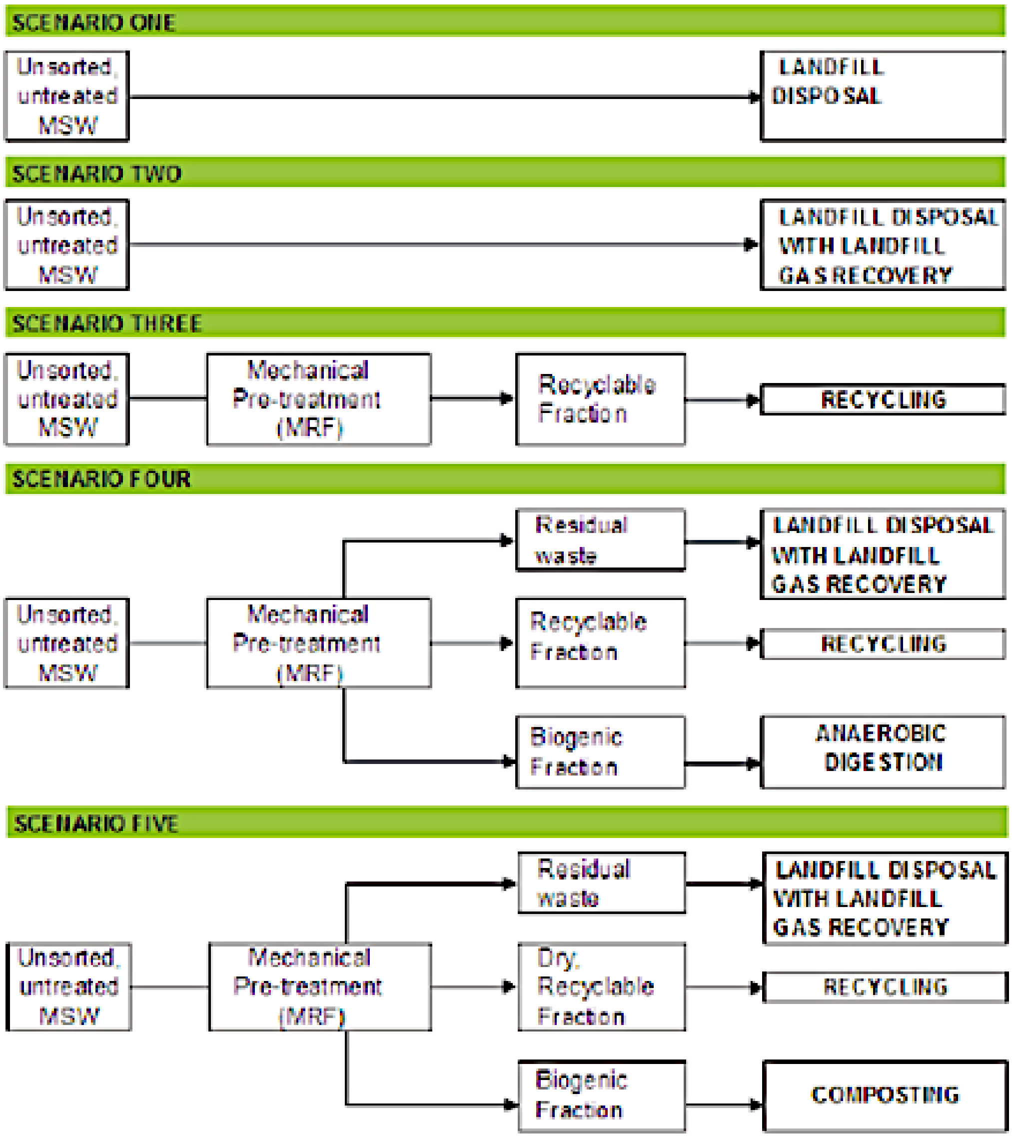


Figure 3.3 - 1: Breakdown of scenarios for WROSE model (Trois and Jagath, 2010)

3.4 Chapter summary

This chapter presented a case study of the major landfills in the eThekwini municipality and indicating the current waste management practices in the area. The raw data pertaining to landfill information was collected and analysed on Microsoft Excel and the results presented as shown. The chapter furthermore provided a short review of a scenario-based modelling software (WROSE model) that can be used to calculate the emission reduction when the AD technology is undertaken on a landfill. This will prove useful when performing assessment for upscaling of the technology. The simulation was not carried out in this work and thus no results or methodology sections were included.

CHAPTER 4 - Methodology

4.1 Introduction

This section presents the methods followed in conducting research and experimentation for this work.

4.2 Materials and Methods

4.2.1 Identification of feedstock

The criteria for identification of a potential feedstock for use in this project was based with emphasis to waste availability, handling, characterizability as well as accessibility. Waste availability with regards to the regularity and stability in waste generation and the handling of waste. These parameters will prove useful when determining the feasibility of utilizing the feedstock to host a biogas facility in the area. The waste must require minimum pre-treatment. Characterizability of waste is made possible through a secure and careful handling of waste with source separation and sorting. This makes it economical to characterize the waste and use it as a feedstock in the Biogas process.

The municipal selected for waste acquisition and investigation was eThekwini Municipality located in KwaZulu-Natal province in the south-eastern part of South Africa. Based on the criteria stated above, a facility was selected where waste is governed by the municipal's metro city council and handled by Durban Solid Waste (DSW) from the area to the landfill. Such a location is Durban Fresh Produce (DFP) market located in 81 flower road, Clairwood. Durban Fresh Produce market handles and facilitates the sales of fruit and vegetables. It is the largest facility of that sort in the region. The facility operates 6 days a week. Of these days, inspection of the food material is done three times a week where fruits and vegetables are assessed and separated. The condemned waste (Food waste declared not suitable for sale by the health inspector) is split between pig feeding (70%) and disposal (30%). The facility has its own transfer station, where waste is compressed in a compactor which is then collected by DSW and sent to Illovo landfill. The condemned waste is relatively clean (not soiled) and requires minimal pre-treatment. The total waste sent to a compactor is composed of condemned waste, packaging waste, sales waste as well as waste collected around the facility. The total volume of a compactor is 15m^3 and it is emptied up to 3 times a day. Thus, making up to 45m^3 of waste sent for landfilling daily. The organic waste ratio to other waste compacted varies according to seasons with summer containing larger volumes of organic waste and the opposite for winter. The inoculum used for this work was Cow dung and was obtained from a farm located in Durban.

4.2.2 Waste Stream Analysis

A stream analysis on the data provided by Clairwood's Fruit and Vegetable Market management was done. This facility contains a variety of the fruits and vegetables that are sold mostly in bulk to businesses. The Fruit and Vegetables are stored in designated cold rooms. For analysis, in this work, common fruits and vegetables were grouped together, for example apple African red and apple blushed gold were simply denoted as 'apple' and the total mass registered. The waste information analysed in this work was for the waste that was deemed unsafe to sell (Condemned food) after a periodical inspection by the health inspector.

4.2.3 Characterisation of Feedstock

4.2.3.1 Waste Sampling and Preparation

The Fruit and Vegetable Market contains a wide variety of fruit and vegetables. The waste was collected, grinded and mixed to attain homogeneity before a sample was taken. A starting mass recorded was about 50kg. Coning and quartering sampling method was used to reduce the sample to a required size. Before characterisation, the sample was milled using a spade and a drum and a laboratory blender to a size less than <1mm. Waste characterisation procedures and methods followed in this work are found in (Walker *et al.*, 2010; Tawona, 2015; Pecorini, Baldi and Iannelli, 2019). A standard protocol for conducting a standardized biomethane potential test was obtained from (Holliger *et al.*, 2016).

<u>Constraints</u>

Due to the Coronavirus pandemic of 2019 (COVID-19), and the civil unrest that occurred in South Africa during the timeline of this work, there was a lot of restrictions and disruptions that occurred. This led to several unfavourable changes that may have affected the results.

- The Substrate was stored for a prolonged period before waste analysis and characterisation and subsequently the bench-run. This was fixed by storing the sample in a cold room at 4 ^{o}C.
- The inoculum (cow-dung) was also stored for close to a week before usage.

4.2.3.2 Ultimate and Proximate Analysis

Ultimate and Proximate analysis of a waste sample was done. Proximate analysis allowed for calculation of the moisture content, ash, and volatile matter. Ultimate analysis provides values of elemental Carbon (C), Hydrogen (H), Nitrogen (N), Oxygen (O) and Sulphur (S). Six (6) samples for the substrate and Inoculum were prepared for this task. These quantities were chosen to improve accuracy of the results.

Total Solids (Dry matter content)

The samples were placed in a crucible of known mass. After recording the initial mass, the sample was dried in an oven overnight at a setting of 105 ^{o}C. The mass was again recorded after the drying process was completed. The Total Solids was thus, the percentage difference between the mass before and after drying.

Total solids % = {(A-B) * 100} / (sample mass, g)

A - weight of dried residue + crucible, g

B – weight of crucible, g

Determination of moisture content

The moisture content was measured using an oven method. The moisture content was calculated as the percentage difference between the Total mass of wet sample and Total Solids content.

Moisture content = 100% - Total solids%

Volatile Solids

The total volatile solids content was calculated as Total Solids minus Ash content. **Ash Content** was calculated through the process of **Ashing**. A dried sample was ignited at 550^{o}C for 2 hours in a muffle furnace, allowed to cool in a desiccator and Ash mass taken using an analytical scale. The percentage difference between the mass of sample before (Total Solids) and after Ashing was calculated and recorded.

Total volatile solids % = {(A-B) * 100} / (sample mass, g)

Ash content % = {(B - C) * 100} / (sample mass, g)

A - weight of dried residue + dish before ignition, g

B – weight of dried residue + dish after ignition, g

C – weight of dish, g

Biodegradability Test

The manometric method described in Robertz (2000) was used in this work for determining the biodegradability of the sample. The manometric method works on the principle that the organic compounds (sample) in the jar are aerobically degraded, resulting in oxidisation to Carbon Dioxide. Carbon Dioxide is then removed from the gas phase by use of potassium hydroxide (KOH) resulting in a pressure drop in the closed system. The pressure drop is directly proportional to the amount of Carbon Dioxide consumed. The progress of the process is thus;

$C(org) + O_2 = CO_2 + salts$

A homogenized sample was placed in 4 jars of 1 litre in size and sealed off. Mass of each sample was maintained at 40g. Connected to the jars was Oxitop pressure sensors to monitor the pressure drop as the oxygen was consumed (Figure 4.2-1). The samples were then placed in an incubator at ambient conditions for 7 days.

Figure 4.2 - 1: A setup for conducting biodegradability test

pH

The pH was measured using a calibrated Thermo-Scientific, Orion pH-meter.

Preparation of Sample

Figure 4.2-2 shows the process followed in the preparation of feed characteristics. The waste collected was crushed to a small size using a (20 by 30cm) spade and a (3.14 m^2)

drum. The comminute was further processed using a 2 litre Kenwood blender to a size <1mm as shown in Figure 4.2-2. This process was necessary to allow a proper mixing of the substrates since a small amount (<10g) was required during a BMP run. The small sized particles are also necessary to maximize surface area for the activity of the various microorganisms. No water or any external liquid was added to the sample to enhance the comminution. The addition can bring about contamination of the sample and thus affect characterisation information.

Figure 4.2 - 2: Collection and comminution of waste to the required size

4.2.4 Biomethane Potential and Biodegradability

Biomethane Potential (BMP) tests measures the biodegradability of organic waste. This test is important for the determination of methane potential of food waste and thus its usability as a potential substrate for Biogas production. As well, the optimization parameters and conditions necessary for best possible outcome. Thus, minimizing possibilities for process failure. Whereas this test is detrimental in Anaerobic Digestion (AD), there exists no standard protocol for conducting BMP. Therefore, the tests are subject to large variations due to non-uniform procedures, equipment, inoculum, and defined heterogeneity (Hansen *et al.*, 2004). This can however be curbed by repeatability and reproducibility tests (Hansen *et al*., 2004). Hollinger et al. (2016) describes the best protocol that ensures reproducibility and repeatability and is useful for conducting comparisons of data from various equipment used. Several appealing methods and procedures are further discussed in various literature (Owen *et al.*, 1979; Hansen *et al.*, 2004; Walker *et al.*, 2010). A simple approach to a BMP test involves fermenting a sizeable amount of waste co-digested with an anaerobic inoculum from which gas volumes and composition are simultaneously measured (Hansen et al., 2004). A simple bench-scale setup is shown in Figure 4.2-3 below and it is made up of a small anaerobic reactor bottle with space above the sample (head space) for gas collection. A sizeable amount of substrate is used in this test, depending on the equipment setup, such that the gas can be contained and measured to the desire of the operator. Risks of pressure build-up inside setups such as that shown in Figure 4.2-3 exist, and close monitoring of the equipment is essential.

South Africa is host to over 700 small scale and commercial digesters (Tiepelt, 2015 cited by (Muvhiiwa *et al.*, 2017). The knowledge base on the subject is limited and legislation supporting the emergence of the technology non-existent. Failure of biodigesters due to lack of first-generation feedstock, associated start up and running costs are some of the challenging factors. Partly due to waste management practices in the country. In some developed countries, commitment by both the government and stakeholders in purchasing the product at a sufficient price and supporting immense research has led to an increase in success of the technology. In Germany, which is one of the leading biogas producing countries, there is a special buying price for electricity from biogas and as well, Germany allows commercial projects from just 400 kW compared to 1 MW in South Africa (Munganga, 2013). Commercialising AD is not the only solution. In India and China, millions of small family sized bio-digesters are used daily for cooking purposes in villages thus reducing demand for grid electricity. This can be applied to a South African setting using the vast feedstock available to the country. This section establishes the best operating conditions for digestion of F&V waste with cattle manure. The experiments were conducted under varying food waste to cattle manure ratios, at mesophilic temperature (36°C) and different pH values. Methane yield for each scenario is reported and a typical substrate and parameters discussed. South Africa can benefit greatly from this study in terms of understanding the best way to get the best out of available feedstock.

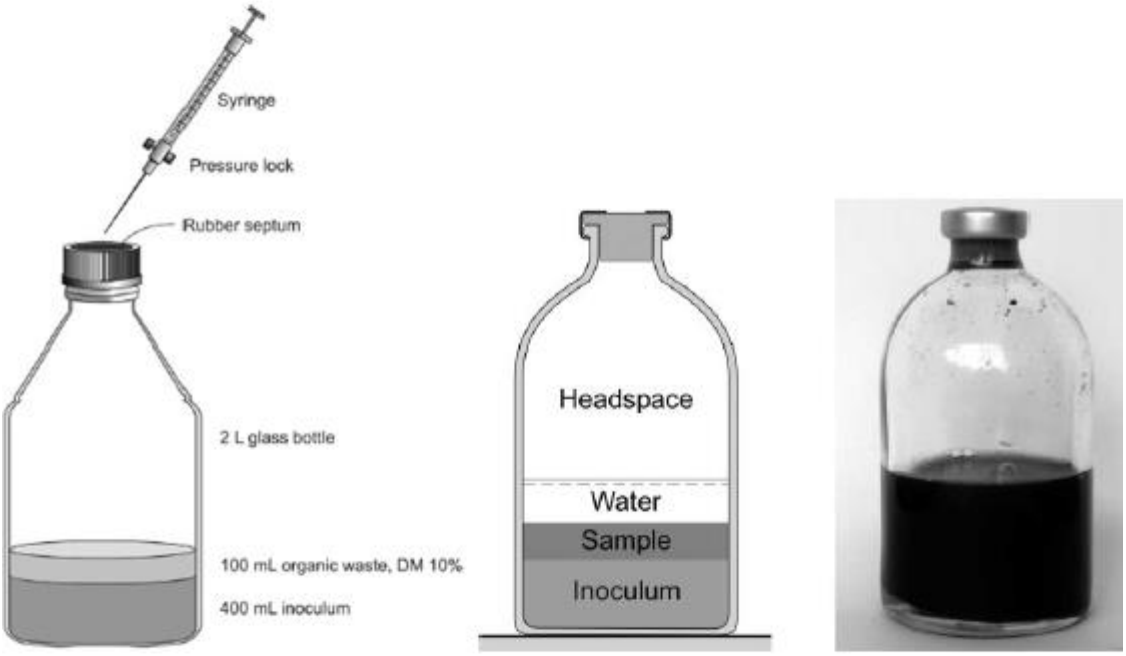


Figure 4.2 - 3: Illustrating an anaerobic reactor and gas sampling (Hansen *et al.*, 2004; Angelidaki *et al.*, 2009)

Several protocols for conducting Biomethane tests, data collection methods and documentation are presented in several articles from various years. Guidelines from the following research were observed in the setting up of the methodology for this work (Walker *et al*., 2010; Angelidaki *et al*., 2009; Hansen *et al*., 2003; *Owen et al*., 1979). Modifications have been made to these to better suit the line of study, working conditions, substrate and inoculum used.

The Substrate used in this work was Fruit and Vegetable waste (F&V) co-digested with cow dung (CD). The equipment setup is shown in Figure 4.2-4 below. The setup was adapted from Tawona (2015) which utilises a displacement method for determining methane volume. The setup has two sections, the reaction side, and the gas volume measurement side. A pipeline connecting each bioreactor with a displacement bottle was erected to allow passage of gas. As the gas enters and bubbles through the alkaline Sodium Hydroxide (3N NaOH) solution to accumulate in the headspace of the displacement bottle, Carbon Dioxide (CO_2) is absorbed leaving behind methane gas (CH_4) which then displaces an equivalent volume to the measuring cylinder below. This setting works on the assumption that only CO_2 and CH_4 are produced anaerobically. Thymol Blue indicator was used to monitor CO_2 concentrations in the displacement bottle.

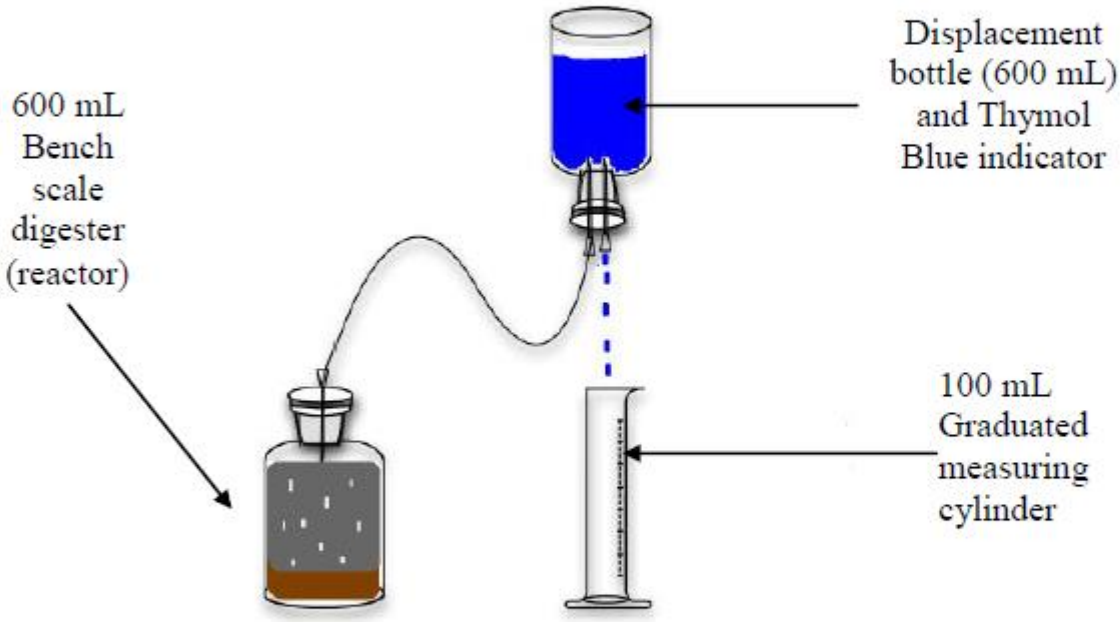


Figure 4.2 - 4: Illustration of equipment setup used in this work (Tawona, 2015)

Equipment Used	Materials Used
600 ml Schott Duran bottles	alkaline Sodium Hydroxide (3N NaOH) solution
Syringe with pressure lock	Thymol Blue indicator
Water bath	N_2 gas
Gas tubing	Distilled water
Geotech GA500 gas analyser	Tap water
Displacement bottle frames	Fruit and Vegetable waste
Shimadzu Gas Chromatogram (GC)	Cow dung
Measuring cylinders	
Shimadzu Gas Chromatography Mass Spectrometry (GCMS)	
Hettich Universal 320R centrifuge	
Mettler Tolero Analytical balance	
Metrohm 913 pH meter	
Lab Pyrolizer	
PerkinElmer frontier FTIR	
PerkinElmer simultaneous thermal analyser TGA	
Energy Dispersion X-ray (EDX) machine	
PerkinElmer CHNS analyser	

Setup Preparation

Water was added to the water-bath and a temperature setting of 36 ^oC was set. An anaerobic atmosphere was created in the bioreactors by purging with N_2 gas for 5 minutes at the beginning of the experiment. There were altogether 12 anaerobic reactors and for each reactor, an assigned displacement setup to measure methane volume. Reactor bottles were modified to allow gas sampling with a gas syringe for GC analysis and pH monitoring. The displacement columns were connected to both the reactors and the measuring cylinder for a direct measurement of displaced volume. Figure 4.2-4 shows the setup arrangement used in this work.

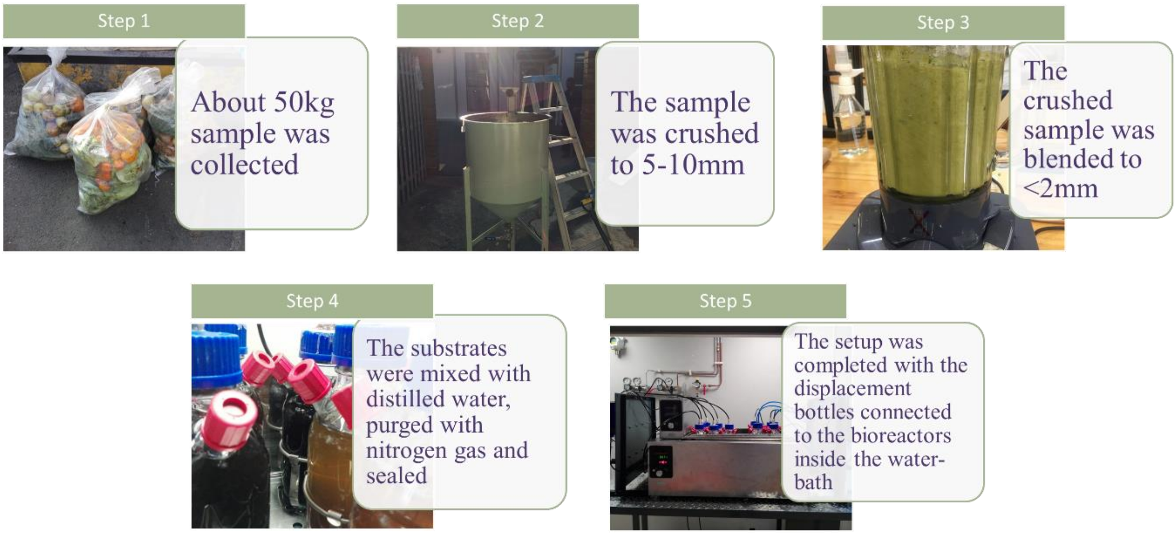


Figure 4.2 - 5: Sample preparation and BMP setup used in this work

Sample preparation

A large heterogenous sample was crushed with a spade in a drum and blended to a required size using a blender. A homogenous mixture was prepared, and a small representative sample was taken for BMP analysis. A small homogenous sample was used to generate manageable biogas volumes that can be contained and measured appropriately with the available equipment.

A reaction sample was made up of inoculum, substrate, and distilled water to reach an effective volume of 580 ml. All reactions were prepared in triplicates for accuracy and in accordance with standards from literature. Gas purging with N_2 was initiated to create an anaerobic environment in the bioreactors. A headspace of 20 ml was maintained in a 600 ml bottle. The loading rate for each arrangement was 1.4g VS/100 ml solution with a co-digestion ratio of CD:F&V varied in the order 80:20 and 60:40. The particle size was at 1-2mm for the 60:40 and 80:20 ratios. Another run with a particle size crushed to below 1mm was done for the 60:40 ratio and was useful for comparative analysis. The pH was varied such that the first triplicate was buffered at solution pH with Na_2HCO3, the second set was not buffered, the third set was adjusted to a pH of 6.5 and buffered with NaK_2PO_4 and $NaKPO_4$ and the last set was a blank containing CD only. The sets were exposed to a mesophilic temperature of 36°C and allowed to stabilize for 2 days before a reading was taken. The digested were mixed daily.

Process Description

Biogas produced in the bioreactors accumulates in the headspace building up enough gas to push into the pipeline. The gas is then directed to the displacement bottle where it bubbles through the NaOH solution into the displacement bottles small headspace. CO_2 is then absorbed into the solution leaving behind CH_4 gas to accumulate into the small headspace and displace an equivalent volume of liquid into the measuring cylinder. The process was observed for a period of 25 days inclusive of the stabilising period. The pH for the buffered and Adjusted pH substrates was not allowed to vary significantly throughout the experiment, however, CD and Non buffered substrates were flexible to vary pH in great margins depending on the activity of the microbials.

Data Collection

Volume displaced into the measuring cylinder was measured with process monitoring for the entire period of experiment. The septa in the bioreactors were constantly observed to notice pressure build-up in the reactors. A sample from the headspace was drawn using a pressure locked syringe and sent to the Gas Chromatogram (GC) machine for analysis. The GC was using a Thermal conductivity Conductor (TCD) detector, a Carbon molecular sieve phase with a packed column (Porapak Q) and the carrier gas used was Hydrogen (H_2). The analysis was 5 minutes long at a column temperature of 35°C and a detector temperature of 250°C. To prevent significant pressure changes in the system, in accordance with the literature, a sample less than 0.7% of the headspace was drawn for analysis. GC analysis provided the ratio of CH_4 to CO_2, and this was used to assess the performance of the various arrangements. The same septa were used to draw out some liquid to measure the pH. At the end of the experiment a Geotech GA500 gas analyser was used to check the methane content in the headspace.

4.2.5 Characterisation of digestate and feasibility analysis

Anaerobic digestion for methane production leads to compositional and chemical changes to the organic feedstock (Möller and Müller, 2012). Studies have shown that only about 15-40% of organic material is converted to biogas and the remaining material is discarded as digestate (Liu *et al.*, 2020). A large amount of digestate is produced annually in biodigesters which prompts the need for proper and sustainable disposal. Valorisation of this residual product is a major milestone to complete circularisation of this technology, with socio-economic benefits. This section is important in the valorisation of digestate as soil conditioners or fertilizers and establishes the benefits of pyrolysis. Research has shown that the benefits of digestate as a biofertilizer are comparable to that of chemical mineral fertilizers and furthermore increases the quality of crops (Panuccio *et al.*, 2019). Depending on the starting material, digestates tend to be nutrient rich in terms of nitrogen (N), phosphorus (P), potassium content (K) and low in Biological Oxygen Demand (BOD) (Tambone *et al.*, 2010; Möller and Müller, 2012; Liu *et al.*, 2020). However, digestate utilisation as fertilizer is often hampered by some negative environmental impacts such as release of pollutants, odours, and presence of heavy metals (Möller and Müller, 2012; Wei, Hong and Ji, 2018; Liu *et al.*, 2020). Some countries have even restricted utilization of digestate in agricultural applications (Taurino *et al.,* 2016 cited in Liu *et al*., 2020). An alternative to this approach is a further treatment of digestate through pyrolysis to form biochar. Biochar is a carbon-rich product from a high temperature treatment (charring) of digestate, a process that reduces the environmental impact of digestates and furthermore increases the economic benefits of AD (Liu *et al*., 2010). During an investigation on cucumbers, digestate application improved the crop's medicinal characteristics by increasing its phenol content (Panuccio *et al*., 2019). Wei *et al.* (2018) reports phenol content in digestates to be 43% after an analysis of pyrolysis products, which was attributed to the high lignin content of digestates.

South Africa is heavily reliant on agriculture for economic performance and sustenance, both commercially and in subsistence farming. Much research has alerted that the soil in the region is flushed of its nutrient capacity, and this negatively affects plant life. This may lead to increased poverty if not tackled in the right manner. Preez, Huyssteen and Mnkeni (2011) stated that 58% of the soils in South Africa have an organic carbon content lower than 0.5% and this is due to bad human practices. Valorisation of digestate for soil improvement can be a major contribution towards poverty eradication, employment creation and green farming.

This section reports the equipment used during the characterisation and post-digestion experimentation conducted on the digestate. The section furthermore states the methods for characterisation of digestates as well as the procedure for pyrolysis of digestates and characterisation of pyrolysis products.

Characterisation of Digestate

The digestate was analysed for pH after AD was completed. Thereafter, the solids and liquids were separated through the work of a Hettich Universal centrifuge. The digestate was then dried for 24 hours in an oven before a dry mass reading was taken. A sample was collected for characterisation of digestate. The tests conducted were VS content, BOD as well as COD tests using the methods described in Chapter 4.

Thermogravimetric analysis was conducted to determine the thermal behaviour of the digestate. Approximately 30mg of sample was heated at a rate of 10^{o}C/min from room temperature to a maximum temperature of 600^{o}C. A Nitrogen gas flow of 20ml/min was maintained. FTIR analysis was done to establish functional groups. Metal analysis was conducted using an Energy Dispersion X-ray (EDX) machine that is equipped with a Scanning Electron Microscope (SEM).

Experimental procedure: Pyrolysis of Digestate

A measured mass of digestate was subjected to pyrolysis in a stainless-steel reactor, equipped with a temperature sensing probe. A water-cooled condenser was connected to the reactor and the outlet connected to a set of three condensate containers. Condensate water temperature was maintained at 5^{o}C and dry ice was placed into the first two containers to condense the volatiles. Prior to the experiment, the reactor and pipelines were purged with N_2 gas to ensure anaerobic conditions. A heating rate of 10 ^{o}C/min was directed to the reactor to heat to the stipulated final temperature. When the reactor temperature reached 80 ^{o}C, dry ice was introduced to the containers. The heavy volatiles condensed into the containers and the rest was removed. After cooling the system, the products, biochar and bio-oil were weighed and stored for further analysis.

Characterisation of Biochar

CHNS analysis was conducted to a sample of biochar. Metal analysis was also done using the EDX equipment.

<u>Characterisation of bio-oil</u>

Characterisation of bio-oil was conducted on a Shimadzu GCMS with a capillary column of dimensions (30m × 0.25mm × 0.25 m). The oil was introduced at an injection volume of 2l. The interface temperature was maintained at 230 ^{o}C. A column flow of 0.8 ml/min at a split ratio of 10 was used. The oven temperature was set to 35 ^{o}C and maintained for 1min, thereafter increased to 80 ^{o}C at a rate of 0.7 ^{o}C/min. The temperature program was then ramped to 105 ^{o}C at a rate of 15 ^{o}C/min and furthermore ramped at 3 ^{o}C/min to 200 ^{o}C where it was maintained for 5 minutes. Helium was the carrier gas.

4.3 Chapter summary

The chapter presented the methods, procedures and tools used during the experimentation stage of this project. The process followed during the anaerobic digestion of the substrates used in this research is presented. Characterisation methods and tools for substrates and digestates are provided. The discussion of the results for the digestion is presented in the next chapter.

CHAPTER 5 – Results and Discussion

5.1 Characterisation of organic waste

5.1.1 Analysis of waste generated at Clairwood market

This section provides the distribution of Fruit and Vegetable waste from 2017 to 2020. The calendar for Clairwood market begins in July and ends in June, the following year. Thus, the charts provided below presents the data such that the second half of the year represents the preceding year and the first half, the following year. During the 2017_2018 calendar, the total tonnage of waste generated at the market was 2400 tonnes (2 400 541 kg). The following year, 2018_2019, around 2800 tonnes (2 806 716 kg) of waste was produced, and 2400 tonnes (2 473 132 kg) for 2019_2020. Depicted in Figure 5.1-1, the market-calendar-year 2018 to 2019 generated the highest amount of waste as compared to 2017-2018 and 2019-2020. Despite the lockdown observed in South Africa in 2020 due to the corona virus pandemic (COVID-19) the year 2019-2020 generated a comparable amount of waste to that of 2017-2018. This was an indication that business in the market was not majorly affected by the lockdown. This was confirmed by the market management.

Figure 5.1 - 1: Amount of waste produced annually at the market

5.1.2 Waste generation on a monthly and weekly basis

The monthly analysis of waste generated is presented in Figure 5.1-2 and Figure 5.1-3. From data presented in Figure 5.1-2, several observations can be made pertaining to the waste generation throughout the year. The first observation is an unbalanced waste generation trend from 2017-2020. Regions of high waste volume and that of a low waste volume are clear. The large amount of waste is produced between December and April, reaching monthly waste tonnage of over 400 tonnes. The low waste region occurs between May and November. A three-year average plot is also shown in Figure 5.1-2 for a better visual of this. In fact, from Figure 5.1-3, it can be observed that over 60% of waste is generated between November and April. The largest amount is recorded in December, February, and March which sums up to over 40% of the waste generated in a year. With the peak contribution in December and March at 14% each, and February with 13%. Several conclusions can be made for this.

- The large amount of waste is because these are summer months and thus high temperatures causes the storage life of materials to be low
- December is a holiday season for many, and thus the demand for fruits and vegetables waste is increased and so does the supply. Because of the rapid increase in the supply, more waste is bound to be generated.
- The quiet months are mostly cold winter months, starting from May. The cold weather allowed for a longer storage life

An average monthly waste generation is around 213 tonnes (213 344 kg). This value represents a range from as low as 109 tonnes (109 273 kg) in July to as high as 363 tonnes (363 600 kg). Another important depiction is the waste disposed of weekly reported in Figure 5.1-4. This data represents the amount of organic waste that can be available as feedstock for AD on a weekly basis. The average weekly waste disposal tonnage was calculated to be around 49 tonnes (49 026 kg). The largest weekly average is found to be 113 tonnes (113 609,4 kg) and the lowest amount recorded is 15 tonnes (15 968 kg). These weekly values are from week 9, “February” and week 28, “July” respectively.

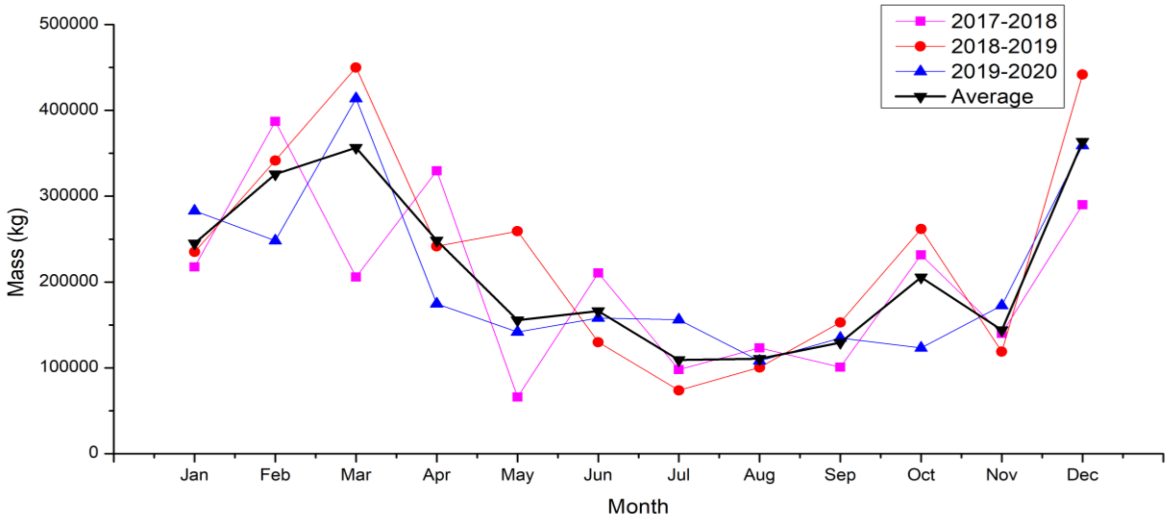


Figure 5.1 - 2: Amount of waste disposed weekly

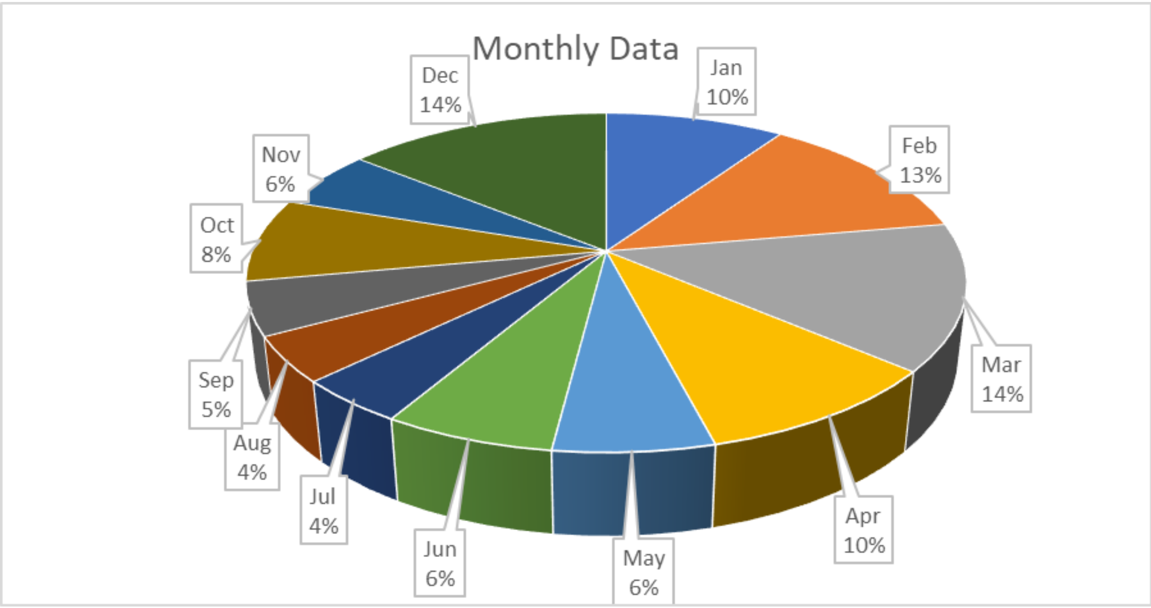


Figure 5.1 - 3: Proportion of waste disposed monthly

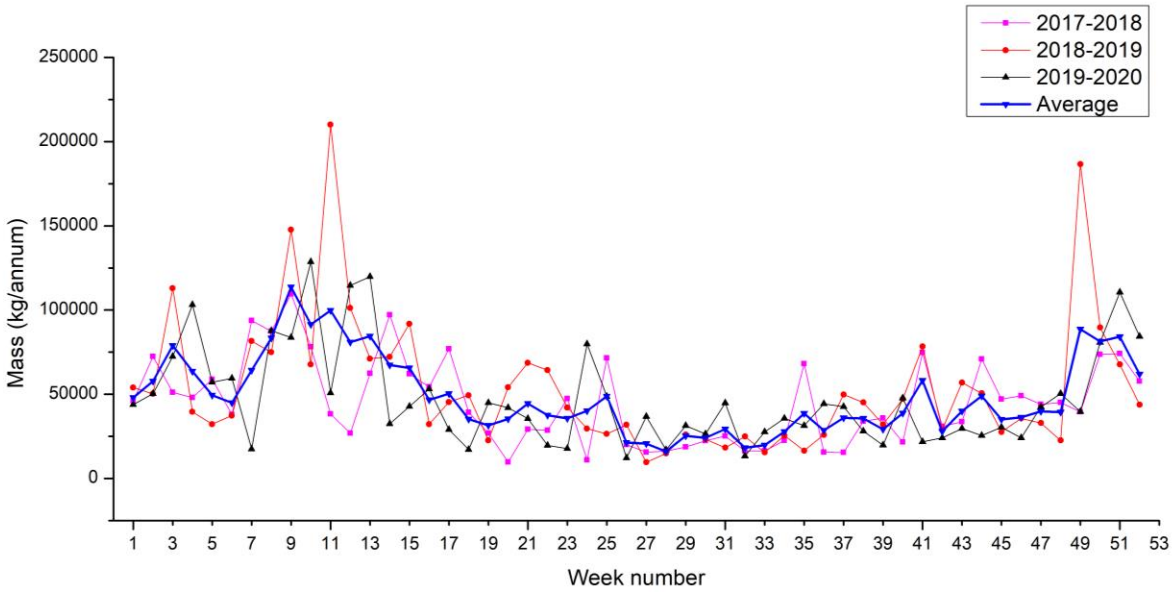


Figure 5.1 - 4: Amount of waste disposed weekly

5.1.3 Analysis of waste by type

The following section discusses the different fruit and vegetables that can be used as a potential substrate for Anaerobic Digestion. This section specifies the waste types as well as the tonnages disposed and provide a comparison of waste streams. It is important to note that only items with a disposal rate greater than 20 tonnes per year (20 000 kg/yr) are reported in this section. A complete breakdown is provided in the Appendices. Figures 5.1-5 and 5.1-6 reports 26 items, however there is altogether 111 food types that are disposed of from the Fruit and Vegetable market. The tonnages provided in this section are averaged to a year and as per weekly disposal, however, the monthly waste generation breakdown for a year provides a better picture of the reliability of these items as a potential feedstock. This information is provided in the next section.

From Figures 5.1-5 and 5.1-6, it is evident that there are certain prominent fruit and vegetables from the protruding spikes in the graphs. Cabbages, melons, and potatoes have the largest disposal rates at the market, compared to the rest. The possible reasons could be that they are the most in demand and hence this is proportionate with the amount that is sold. The other reason could be that they easily rot thus requiring a much shorter storage period. Other important observations are outlined below;

- From Table 5.1-1, the percentage disposal rate of melons (proportioned to the 26 items) was 15%, followed closely by Cabbage with 14%. Potatoes and Onions contribute a mass disposal percentage of 10% and 9% respectively.

- There are 6 waste types (items) that generates over 100 tonnes (100 000 kg) of waste per year. These include, melons (15%), cabbages (14%), potatoes (10%), onions (9%), lemons (5%) and oranges (5%).
- The next range above 50 tonnes (50 000 kg), includes peppers (4%), spinach (4%), californian salad (4%), broccoli (3%), carrots (4%), cauliflowers (3%), butternuts (3%) and apples (3%).
- Cabbage has the highest average weekly tonnage of 8760 kg followed by melons with 7859 kg.

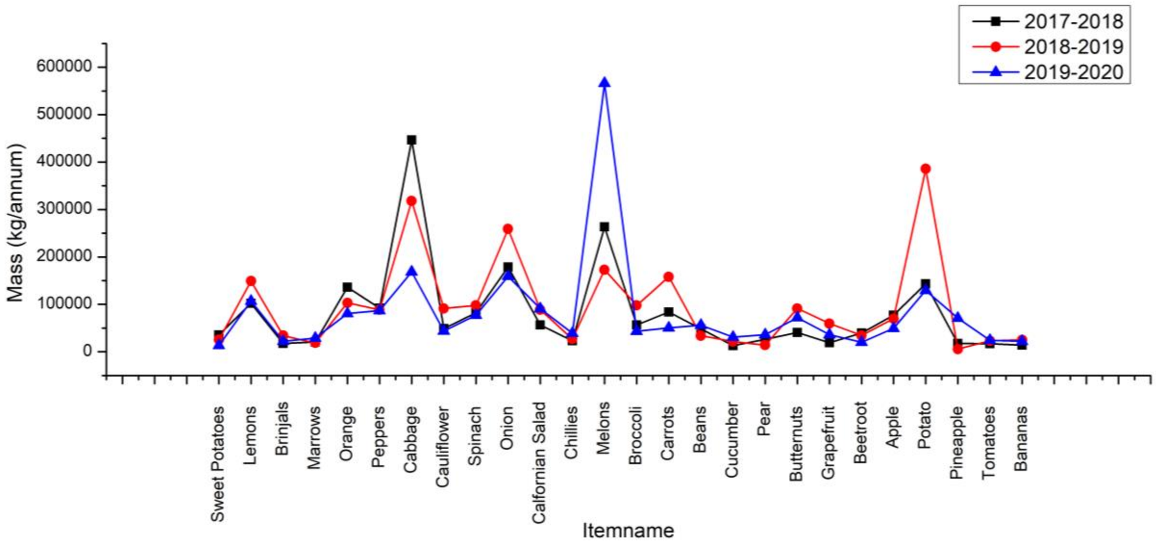


Figure 5.1 - 5: Total mass per year per waste item

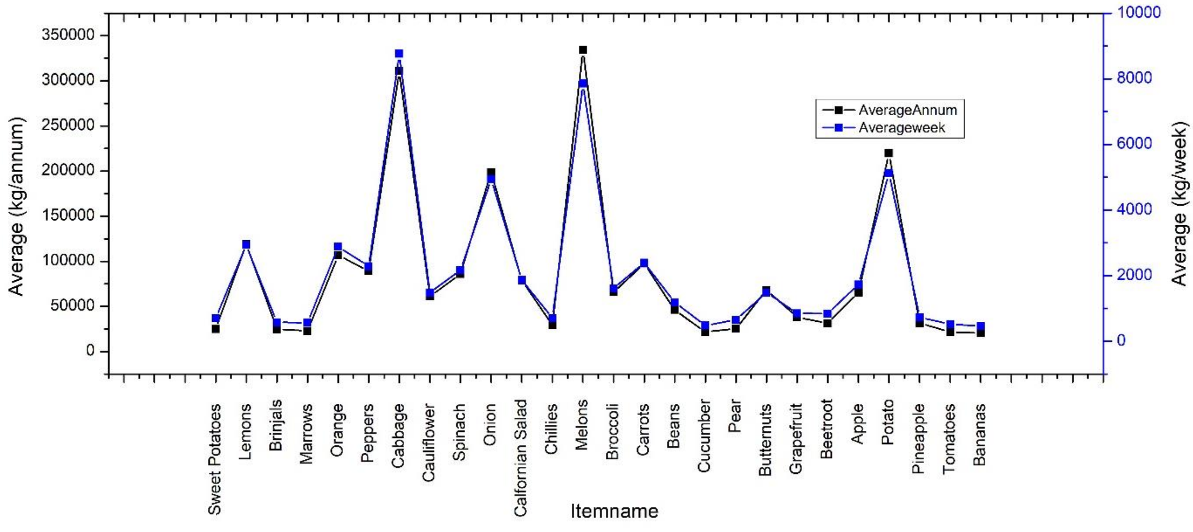


Figure 5.1 - 6: Comparison of weekly and yearly averages of F&V waste

Table 5.1 - 1: Waste stream analysis

Item Name	Average per Annum	Average per week	Proportion %
Sweet Potatoes	24856	698	1
Lemons	119386	2939	5
Brinjals	24677	580	1
Marrows	22849	565	1
Orange	106811	2886	5
Peppers	89155	2289	4
Cabbage	311122	8760	14
Cauliflower	61327	1485	3
Spinach	85928	2169	4
Onion	198890	4945	9
Californian Salad	78886	1864	4
Chillies	29719	709	1
Melons	334143	7859	15
Broccoli	65937,5	1609	3
Carrots	97570,16667	2395	4
Beans	46127,66667	1185	2
Cucumber	21752,8	481	1
Pear	25500,5	658	1
Butternuts	68198,73333	1488	3
Grapefruit	38126,5	853	2
Beetroot	31302,66667	846	1
Apple	65345,03333	1732	3
Potato	219556,3333	5123	10
Pineapple	31478,33333	723	1
Tomatoes	21586,83333	524	1
Bananas	20455,33333	463	1

5.1.4 Defining a typical sample

From the information presented in the previous section, it is clear that a feedstock collected from the market will comprise a larger proportion of cabbage, melons, onions, potatoes, lemons, carrots and oranges. This section thus provides more detail into these fruits and vegetables. The section specifies monthly breakdown data as well as methane potential for the items. The importance of this is that, for a substrate to be used a feedstock in AD, there must be a good understanding of the consistency in availability as well as the expected produce. This information will aid in construction of a suitable reactor as well as mitigation measures that may be required given the foreknowledge on the substrate provided. Figures 5.1-7, 5.1-8 and 5.1-9, summarises the monthly disposal rates of the aforementioned fruits and vegetables. For better clarity, the graphical representation was done in two plots, separating the fruits and vegetables by tonnage disposed. The third figure provides a cumulative frequency report of the monthly waste per item, further defining the disparity between waste data for various wastes. Table 5.1-2 then tabulates the typical methane potential of the fruits and vegetables discussed. In Figure 5.1-7, items with a disposal rate greater than 25 tonnes per month (25 000 kg/month) were presented and the rest were placed in Figure 5.1-8. The monthly item analysis is thus:

Cabbage

The average disposal rate for cabbage at the market is 311 tonnes per year (311 122 kg/year). Figure 5.2-7 displays the monthly distribution of cabbage waste from a three-year period ranging from 2017 to 2020. Cabbage waste generation trend is evenly spread across a year. This is made clear in the cumulative chart in Figure 5.2-9, which indicates an almost straight line (constant increase) for cabbage. Much of the disposal is done in December with about 50 tonnes (50 860 kg), followed by October with 41 tonnes (41 672 kg). The least amount of waste generated is found in the middle of the year between June and September where disposal is under 20 tonnes. The average storage length of cabbage at the market was 14 days. As a feedstock for Anaerobic Digestion, it can produce about 0.309 l/g VS of methane (Gunaseelan, 2004).

Melons

Melons generates the most vegetable waste with 334 143 kg per year. Most melons are seasonal and are harvested in the summer. However, some melon types are available in other parts of the year as presented in Figure 5.2-7. During the peak season, which is graphically presented from December to March, disposal rates are well over 40 tonnes. The disposal rates reach up to 88 tonnes per month (88 469 kg/month) in March and 77 tonnes (77 585 kg) in February. A very sharp increase in the cumulative chart during these months affirms this observation. In the mid-year the disposal is below 20 tonnes. The length in storage at the market was calculated to be 13 days. A typical methane potential for melons is 1.254 l/g VS and the biogas produced is reported to have a high flammability (Hussaini *et al.*, 2021).

Potatoes

In a year, a total amount of potato waste produced is 220 tonnes (219 556 kg). Most potato waste is generated in March. About 43 % (94 43636 kg) of potato waste is generated in this month. Only in months between February and June as well as December is potato waste generation surpassing 20 tonnes. 86% of annual potato waste is produced between February and June. Otherwise, the waste generation is as low as zero in July, following a three-year average. This is visualized in the cumulative graph, Figure 5.2-9, where it is evident that there is not much change in June and October in potato waste. The storage rate of fresh potatoes at the market before disposal is 10 days. Methane production from potato waste can be as high as 0.426 l/g VS (Stewart, Bogue and Barger, 1984).

Onion

Onion waste generation is unevenly distributed throughout the year. Of the 199 tonnes (198 890 kg) of onions disposed, over 73% is generated in the first 4 months of the year. Including December, this makes it 84%. Between May and November, only 16% of the waste is produced, inclusive of less than 1% in the months of May and September. During these months the waste generated is under 11 tonnes. Onions are in storage, on average up to 14 days. A typical methane potential is 0.212 l/g VS (Ligisan and Tuates, 2016).

Carrots

The waste distribution for carrots in a year is uneven, and the most waste generation is found from October to March. 88% of carrot waste is produced in this period. The peak waste generation is in December where 20% of the 97 tonnes (97 570 kg) is accounted for in this month. Between April and September, waste generation is below 6 tonnes per month. This is made clearer in the cumulative chart, Figure 5.2-9, where the curve seems to flatten, thus indicating little to no increase in quantity in that period. The storage life of a carrot at the market was 16 days. A typical methane potential of carrots is 0.309 l/g VS. (Gunaseelang, 2004).

Lemons

There is on average, 119 tonnes (119 386 kg) of lemons waste produced per year at the fruit and vegetable market. However, most of the waste generation is found in the last 4 months of the year, which accounts for about 52% of the waste. With the exception to June, the waste generation rate is below 10 tonnes per month between January and August. The storage length of lemons was found to be 35 days, the highest amongst the fruit and vegetable wastes discussed in this section. A typical methane production from lemons is 0.473 l/g VS (Gunaseelang, 2004).

Oranges

Annual waste generation for oranges is 106 tonnes (106 811 kg). The highest average leans towards the middle of the year as compared to the beginning and the end of the year. 48% of waste is generated between June and August, the winter months. The peak month for oranges is June where 22% (24 tonnes) of the waste was generated. The average storage length for oranges was 17 days before disposal. A typical methane potential for oranges is 0.473 l/g VS (Gunaseelang, 2004).

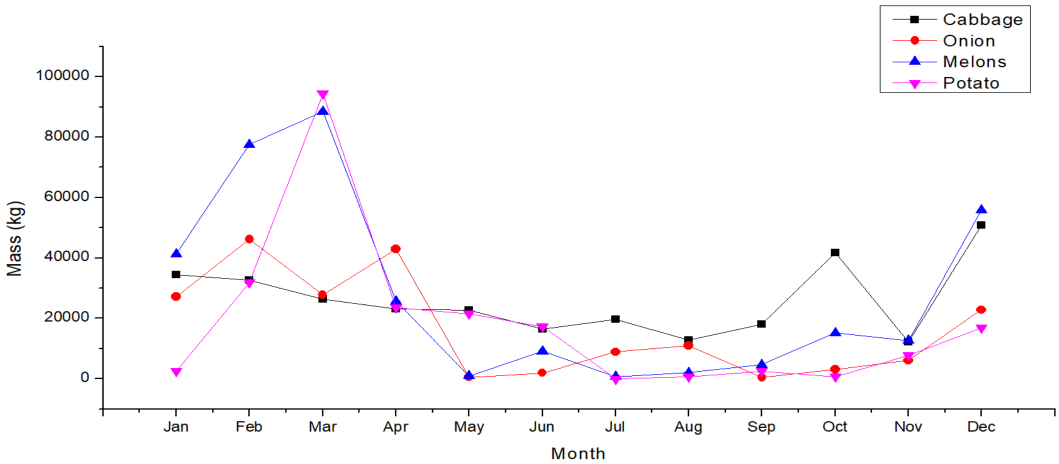


Figure 5.1 - 7: Monthly disposal rates of different fruits and vegetable waste

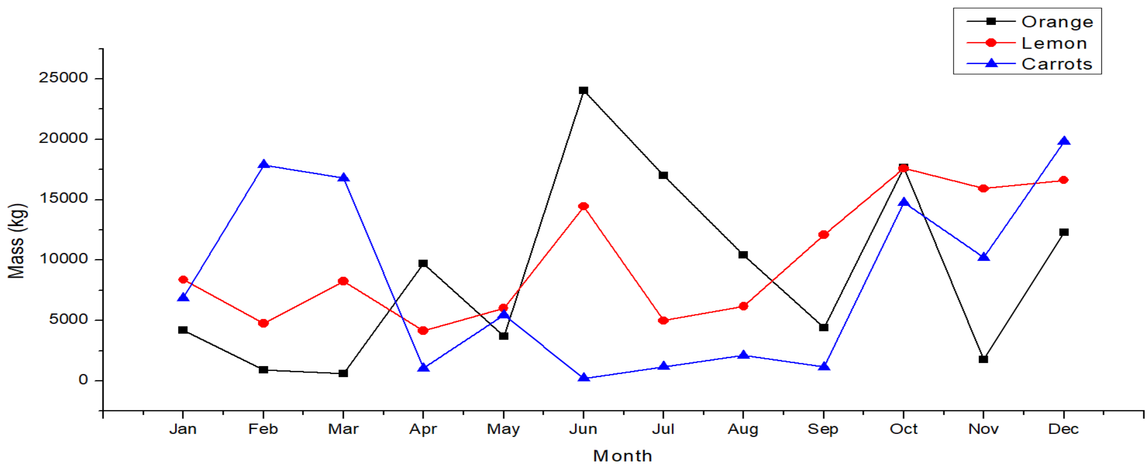


Figure 5.1 - 8: Monthly disposal rates of different fruits and vegetable waste

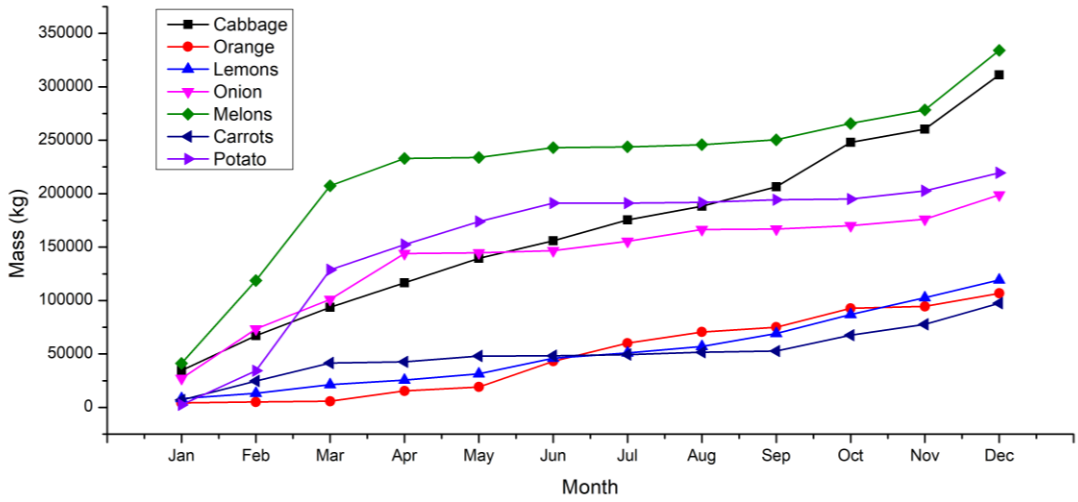


Figure 5.1 - 9: Cumulative frequency chart for monthly disposal rates of different fruits and vegetable waste

Table 5.1 - 2: Typical methane yield for fruits and vegetables in this sturdy

Item	**Total Solids %**	**Methane yield l/g VS**	**Cross reference**
Cabbage Stem/leaves	91.2/91.8	0.309-0.291	(Gunaseelang, 2004)
Potato peels	90.9	0.267	(Gunaseelan, 2004; Ward *et al.*, 2008)
		0.426	(Stewart, Bogue and Barger, 1984)
Carrot Leaves/petiole	93.1/91.3	0.241-0.309	(Gunaseelang, 2004)
Orange peels/pressings	94.7/92.3	0.455-0.502	(Gunaseelang, 2004)
		0.473	(Hussaini *et al.*, 2021)
Lemon	96.8	0.473	(Gunaseelang, 2003; Ward et al., 2008)
Melons		1.254	(Hussaini et al., 2021)
Onion		0.212-0.4	(Ligisan and Tuates, 2016; Ji *et al.*, 2017)

5.2 Proximate and Ultimate Analysis results

This section discusses the proximate and ultimate analysis data used in this work. This information is displayed in Table 5.2-1 below.

Table 5.2 - 1: Characterisation of input material

Description	Food Waste (FW)	Cow Dung (CD) Dry	Cow Dung (CD) Fresh
Moisture Content %	87.5 ± 0.012	42.44 ± 1.09	80.84 ± 0.32
Total Solids TS %	12.5 ± 0.126	57.56 ± 1.09	19.16 ± 0.32
Volatile Solids VS %	92.5 ± 0.031	56.01 ± 0.32	79.94 ± 0.29
Ash content %	7.5 ± 0.031	43.99 ± 0.32	20.06 ± 0.29
Fixed Carbon (FC)	31.5	18.67	
Calorific Value MJ/kg	16.11	7.55	16.6
Carbon (C) %	40.31 ± 1.27	34.21 ± 6.94	
Hydrogen (H) %	6.42 ± 0.250	5.36 ± 1.83	
Nitrogen (N) %	2.67 ± 0.49	2.75 ± 1.32	
Sulphur (S) %	0.48 ± 0.092	0.595 ± 0.30	
[a]Oxygen (O) %	42.64	37.10	
[b]Proteins %	16.69	17.19	
C/N ratio	17.76	12.44	
	Carbohydrates analysis		
Arabinose %	1.098 ± 0.54	0.51	
Galactose %	1.94 ± 1.10	0.54 ± 0.14	
Glucose %	43.81 ± 13.63	11.27 ± 0.07	
Xylose %	3.02 ± 0.40	6.14 ± 0.01	
Mannose %	1.37 ± 0.66	0.34 ± 0.03	
Sum	51.23	18.8	
	Reactivity		
Ri7 mg/L	18.7	19.1	

	pH analysis	
pH	4.7 ± 0.02	9.67 ± 0.23

[a] calculated by difference method [b] calculated by 6.25*N

Waste characterisation information is displayed on Table 5.2-1 above. A comparison of fresh and dry cow dung (old manure) was done to establish the difference in properties in order to assess the effect these can have on the process of AD as inoculants. Old manure is commonly used as fertilizer because of the high nutrient to dry mass content. A lot of moisture is lost with time as the cattle trample over it and is soiled in the ground. However, as will be presented in this section, the elemental content does not vary much from its fresh state. A moisture content of 43% was measured for dry CD compared to fresh cow dung which contained a moisture content of 81%. Over 56% of the 57% TS for dry cow dung was VS in comparison to 80% of the 19% TS for wet cow dung. This represents over double the amount of VS available per gram of dry cow dung (0.32g VS/g) compared to wet cow dung (0.15 g VS/g). However, this may not be translated to the amount of nutrients available for activity of microorganisms or the efficiency in utilization of the nutrients. Elemental analysis with the exception to the carbon content fell within 2% for both cow dung.

Fruit and vegetables are typically rich in moisture. The moisture content of fruit and vegetable waste (F&V) used in this work was 87.5%, leaving behind just above 10% of a TS mass to supply the volatile mass. A VS content of 92.5 % was obtained for F&V, this translated to 0.12 g VS/g sample, a value lower than that of either CD considered for this work. Ash content was much higher for CD in comparison to F&V waste. A high ash content is associated with a high nutrient level (Tawona, 2015), and is mostly utilised in composting or as soil conditioners (Gustavsson *et al*., 2014 cited in Tawona, 2015), a common application for cow dung. However, for the same reasons, manure has emerged as a good material for AD processes. In co-digestion or as a single feed which is common in farm digesters.

A C:N ratio was found to be 18:1 for F&V waste whilst cattle manure was lower at 12.4 %. This ratio is important in the efficient nutrient uptake of microbes and as well the methane potential of the substrates. Co-digestion is usually employed to regulate the ratio to ensure a good C:N content however, for the case of F&V waste with CD used in this work, the resulting C:N ratio was still low which prompts a possible need for adding of a nitrogen rich material. The optimal C:N ratio for AD is in the range of 20-30:1 whilst a higher nitrogen content risk accumulation of ammonia which hampers methane production. To determine the C and N contents, elemental analysis was conducted. Elemental analysis is important in determining the usability of food waste. CHNO analysis can be used to estimate biogas and methane potential of a substrate. This is described in chapter 2 and is demonstrated later in this section. CD was found to be rich in nitrogen but have a low carbon content whilst F&V waste is rich in carbon, hydrogen and oxygen but have a low nitrogen content. Sulphur analysis showed that the sulphur content of F&V waste and CD was 0.41% and 0.81% respectively. A high Sulphur content is not desirable as it risks formation of a pollutant H_2S inside a reactor leading to increased foul odour and inhibition to the AD process (Yang *et al.*, 2015). High concentrations of H_2S furthermore raises health risks in a case of gas release and increases corrosion potential of the gas inside pipes (Okoro and Sun, 2019). It is normally in the range of 0.1 to 3% (Montebello et al., 2012 cited by Yang et al., 2018). The emergence of this gas is highest at a low pH range and reduces exponentially after a pH of 6, reaching minimal values after a pH of 8 upwards (Möller and Müller, 2012; Yang *et al.*, 2015; Okoro and Sun, 2019).

Another indicator for digestibility of organic material is the carbohydrates content. F&V waste was found to be rich in carbohydrates as compared to CD. A similar observation was obtained by (Sawyerr, 2019) and Tawona (2015). High carbohydrates materials tend to have a higher biogas potential. From a study by Yang et al. (2015), the removal of carbohydrates during the digestion process was coupled with an exponential increase in biogas production. This meant that a high removal efficiency of carbohydrates is desired for a successful digestion process. F&V waste had over 50% carbohydrates compared to 18% for CD. Both F&V and CD are rich in glucose content compared to other carbohydrates with 43% and 11.27% for food waste and cow manure respectively. The protein content for F&V waste was 16.69% and 17.19% for CD. From a similar analysis by Tawona (2015), F&V had a protein content of 16.3 g/100mg as compared to 11.5 g/100mg for CD. A negligible starch <0.1 g/100g was obtained compared to 1.7 g/100g for F&V waste. Just like carbohydrates, proteins are desired for a good yield in biogas. A steady increase in biogas was observed with a steady increase in proteins (Yang et al., 2015). Fat content was found to be 3.4 g/mg and 0.5 g/mg. Fats degradation has the highest biogas and methane potential compared to proteins and carbohydrates (Baserga 1998 cited in Tawona, 2015). The content of proteins, carbohydrates and fats can be used to estimate methane potential for organic waste.

F&V waste contains a great deal of highly acidic citrus fruits which lowers the overall pH of the substrates. This is a major reason why co-digestion is employed when dealing with F&V waste to counter the pH for an optimal process. For this work, a 4.7 pH for food waste was countered by a 9.67 pH for cow dung.

F&V waste and CD are high calorific value materials. This increases their usability to cover a wide range of energy recovery processes such as pyrolysis, incineration, and gasification. In extension, their digestate material after digestion. The energy potential for wet cow dung and F&V waste was higher than 16 MJ/kg. However, dry cow dung had a calorific value of 7.55 MJ/kg which was less than half the calorific value of fresh cow dung (16.6 MJ/kg) and F&V waste (16.11MJ/kg). The high heating value of these substrates was associated with their lignin content (L) by (Acar and Ayanoglu, 2012) who proposed the following correlation:

Heating value = 0.0979L + 16.292

5.3 Fourier Transform Infrared Spectroscopy (FTIR) analysis of substrates

Functional group characterisation was conducted on the substrates to further identify them. An FTIR spectra for both substrates showed very similar peaks as outlined in Figure 5.3-1 below. The broad peaks at 3276 cm^{-1} and 3296 cm^{-1} for F&V waste and CD, respectively indicates single bonds containing O-H stretching vibration, which is associated with the presence of phenols, alcohols, or carboxylic acids (Larkin, 2011; Bureau, Cozzolino and Clark, 2019). Small peaks at 2920-2918 cm^{-1} were attributed to the asymmetric and symmetric CH_2 bonds, whereas the peaks vibrating at 1634-1633 cm^{-1} were a C=O group which is either a carboxylic acid/ester or an aldehyde/ketone group. The absorption bands between 1027 cm^{-1} and 1052 cm^{-1} were allocated to the C-O and/ or C-C stretching vibrations. The small peaks ranging from 1147 cm^{-1} to 1422 cm^{-1} indicated the presence of a C-N stretch associated with aromatic and/or aliphatic amines. The increase in craggy peaks in this range indicated the increase in free sugar concentration and an increase in soluble solids (Bureau, Cozzolino and Clark, 2019). No peaks were identified at 2300 – 2200 cm^{-1} indicating the lack of triple bonded CC, CN compounds in the substrates.

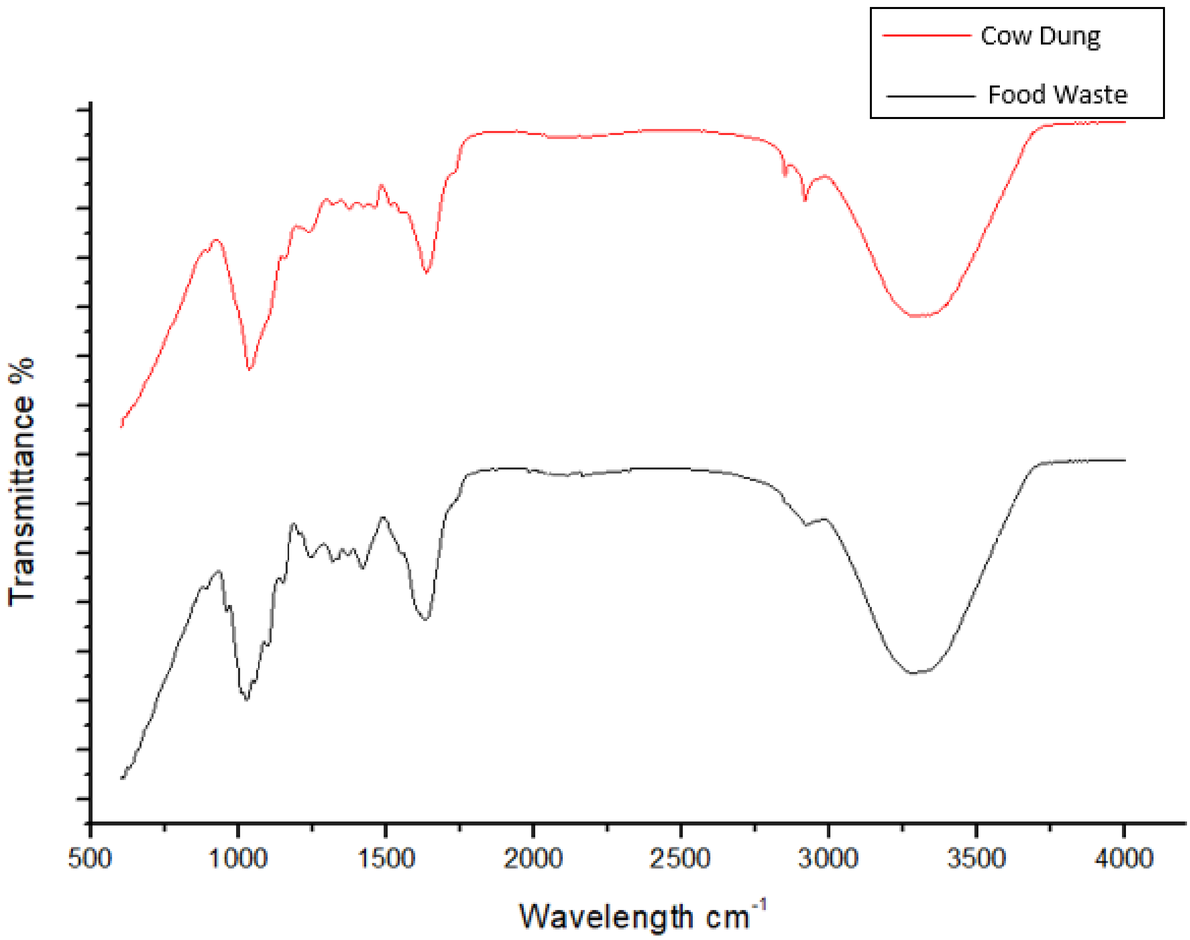


Figure 5.3 - 1: FTIR spectra for fruit and vegetable waste and cow dung

5.4 Biomethane Potential and Biodegradability

This section discusses the results from the co-digestion of F&V waste and CD. A loading rate of 1.4 g/ml VS in a 580ml solution was used. The loading rate was sufficient to ensure proper collection and handling of the biogas with the available equipment. Thus, the mass of substrate used was calculated based on the volatile solids of the substrates which is calculated in Chapter 4. The results to be discussed were optimized in terms of pH and particle size. However, the C/N ratio for the substrates as discussed was below the optimal range of 20:30. The C/N ratio of F&V and CD substrates were found to be 17.76 and 12.44 respectively. Thus, an additional carbon source was recommended.

5.4.1 pH analysis

An important parameter investigated in this work was the effect of pH on the process performance using methane generation in a 25-day period as a performance indicator. As discussed in section 5.2, F&V substrates are very acidic, and owing to the low TS and high VS they are quickly hydrolysed leading to a rapid drop in pH in the early stages of the AD process (Ji *et al.*, 2017). The rapid drop in pH leads to inhibition of methanogenic bacteria and thus decreasing or ceasing methane production in just a few days. A solution to this problem was the introduction of a co-digestion substrate CD, which is highly alkaline. CD furthermore acted as a source of inoculants necessary for fermenting F&V biomass. Thus, providing for a better process stability and efficiency. Otherwise, an alkalinity agent would have been required (Perin *et al.*, 2020). Other inhibitors are associated with pH as well; NH_3 is most inhibiting at pH >7 whereas VFAs and H_2S inhibits the AD process at pH <7 (Ji *et al.,* 2017).

Optimality in starting pH, for a one stage batch process for F&V substrates is found in the range of 6.5 to 8. Whilst it is common that the most optimality is achieved in the upper optimal region (at pH above 7), several studies obtained the highest methane production at a pH of 6.5 (Sibiya, Muzenda and Tesfagiorgis, 2014; Mao *et al.*, 2017). Sibiya, Muzenda and Tesfagiorgis (2014) conducted AD of grass silage in co-digestion with CD and obtained a higher average methane yield for a pH of 6.5 compared to 7.2 at temperatures 35^{o}C, 40^{o}C, 45^{o}C and 50^{o}C. Thus, defining the best working pH for a process varies between substrates.

Table 5.4-1 presents the starting and ending pH of the substrates used in this work. As already mentioned, the reactors were set to different pH levels (6.5 and buffered, buffered at initial substrate pH and the last set unbuffered). The starting pH of the substrate mixture was 9.26 and 8.41 before and after addition of a buffer, respectively. A low pH was achieved through the addition of HCl. An unbuffered run provided a base level for the co-digestion substrates under undisturbed conditions and is favourable for small, affordable, homestead anaerobic digestors. For every run, a mono-digestion triplicate set of CD was included. Due to the small volume of the reactors, repetitive pH measurements during the commencement of the process were not done to prevent drastic changes that may affect optimality inside the system and thus affecting microbial action. However, an observation that can be made from close-up view in Figure 5.1-3, there were some biodigesters that turned dark in colour during the process. This was not directly linked to process performance since a decline or surge in methane generation rate was not observed in comparison. However, the reactors with a darker solution had a very high pH above 8. Despite the high pH, a dark colour of the digestate is often characteristic of a well digested sludge (Wisconsin Department of Natural

Resources, 1992). This was characteristic of unbuffered digesters. This is demonstrated in Table 5.4-1 below for unbuffered reactors and CD only (also unbuffered). A very high ending pH for these was observed. There was no specific pH for the various unbuffered triplicate reactors observed as the pH could be as low as 6.8 and as high as 12.7. Buffering was a success as the buffers were able to maintain the pH for the reactors throughout the experiment.

Table 5.4 - 1: pH for the solutions at the beginning and the end of the runs

Buffer	**No Buffer**	**Adjusted pH**	**Cow Dung only**
		Starting pH	
8.41	9.26	6.51	9.54
		Ending pH	
7.41 +/- 0.16	9.85 +/- 4.23	6.72 +/- 0.03	10.97 +/- 2.68

5.4.2 Influence of pH on methane yield

Figure 5.4-1 presents the results obtained from co-digestion of F&V waste and CD under pH conditions described above. In observing the methane yield for same substrates (same pre-treatment and co-digestion ratio), the buffered reactors produced the highest methane compared to unbuffered and adjusted pH. Reduction in methane yield between Buffered and Unbuffered reactions were in the order of 30%, 73% and 15% for 60:40 (<1mm), 60:40 (1-2mm) and 80:20 (1-2mm) respectively. Unbuffered 80:20 (1-2mm) reactors compared well with its buffered reactors with only just a 15% reduction. This is an important observation which shows the capacity of this mixture to operate well in the absence of external chemical buffer. A cheaper alternative to the buffered mixtures, and very realistic for non-commercial applications. This must have been influenced by the large presence of a highly alkaline CD compared to F&V substrate. A 60:40 ratio contains a larger amount of F&V substrates which as discussed are very volatile with a low solids content and very acidic. They are prone to inhibition. Such an occurrence was the reason for a very low methane yield for the unbuffered 60:40 (1-2mm) triplicate set. As discussed above, referencing to Figure 5.4-3 below, 60:40 (1-2mm) generated a very high methane yield in the first 3-5 days comparably before the graph flattened due to process inhibition. This makes this ratio the least favourite to use in the absence of a buffer or proper pre-treatment. With the exception to the inhibited 60:40 (1-2mm) run, adjusted pH generated the least methane. Thus, indicating the substrate compatibility to a higher optimality (pH>7). A drop in methane yield in comparison to the buffered reactors was in the order of 42%, 69% and 56% for 60:40(<1mm), 60:40 (1-2mm) and 80:20 (1-2mm) respectively. Compared to the unbuffered reactors, adjusted pH triplicates for the 60:40 (<1mm) resulted in a 12% decline in methane yield.

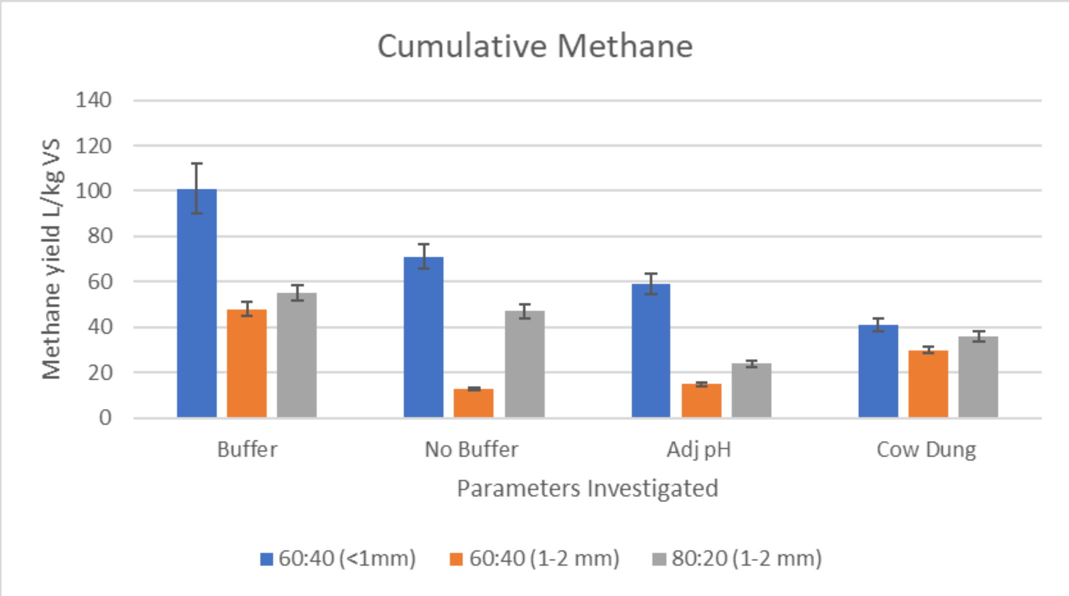


Figure 5.4 - 1: Cumulative methane yield (L/kg VS)

5.4.3 Co-digestion ratio

Co-digestion ratio influences methane generation. Under same pre-treatment conditions, 80:20 co-digestion ratio produced a higher methane yield compared to 60:40 co-digestion ratio under every scenario investigated. However, the difference was not significantly large under buffered conditions. Based on the total methane yield after 25 days, 60:40 was 13% lower under buffered conditions and 38% for adjusted pH. A better look at these is observed on the cumulative graphs, Figure 5.4-2, and Figure 5.4-4 for buffered and adjusted pH respectively. Under buffered conditions, a nearly consistent gap between the 80:20 and 60:40 graphs was observed. Both plots assume a nearly linear increase in methane yield which is not very characteristic of a complete digestion plot. This is displayed in Figure 5.4-2. This meant that the process was still expected to produce more methane over a much longer period. The result was rather associated with a poor pre-treatment. Adjusted pH plot for both ratios were about the same for the first 15 days before the 60:40 methane generation rate fell out. The 60:40 ratio under these conditions was characteristic of the AD graph, which means that the rate of methane generation was approaching zero. This shows that digestion of this substrate at a low pH generates a far less methane yield. The 80:20 ratio was however, still expected to increase. For unbuffered conditions, as already stated, the 60:40 ratio experienced inhibition early in the run which subsequently affected methane generation (Figure 5.4-3). Despite methane generation not ceasing, the rate never increased to expected rates but as the graph showed, the rate of increase was approaching zero only after 21 days. The 80:20 ratio was however uninhibited and assumed a graph similar to that of a buffered mixture despite a slightly lesser yield. The almost linear plot shows that methane generation was still expected to increase over a longer period.

5.4.4 Pre-treatment of substrates

Pre-treatment of substrates is one of the most important steps in AD. In this research, it was found out that it is furthermore more detrimental to AD success than either co-digestion ratio or pH investigated, in reactor performance. A high performance was observed for substrates pre-treated to a size <1mm compared to either of the substrates pre-treated to a size range 1-2 mm. This was true for daily methane production and total methane yield at the end of the run. For the same running period, a smaller sized feed (<1mm) was almost completely digested as compared to the other substrates which were still picking in methane generation by day 25. The cumulative methane generated at day 25 was almost double that of larger particle sizes for Buffered reactors. Regardless of the co-digestion ratio. A methane yield of 101 L/kg VS was found compared to 48 L/kg VS and 55 L/kg VS for 60:40 (1-2mm) and 80:20 (1-2mm) respectively (Figure 5.4-1). Unlike for the unbuffered 60:40 (1-2mm) ratio, the 60:40 (<1mm) ratio did not experience any inhibition. Which meant that inhibition could be associated more with particle size to microbial activity than co-digestion ratio and pH. The same observation is true for adjusted pH mixture. Regardless of pH, the 60:40 (<1mm) produced the highest daily and total methane compared to the 80:20 (1-2mm) and the 60:40(1-2mm). The least methane produced was 59 L/kg VS, at a low pH of 6.5, which was still higher than the highest methane generated by the 80:20 (1-2mm) and the 60:40 (1-2mm) by the 25th day under a buffered state. All graphs for the 60:40 (1-2mm) shows a characteristics AD plot, showing a near maximum methane yield under 25 days. This is a shorter reactor residence time with a much higher methane yield, representative of a good digestion rate.

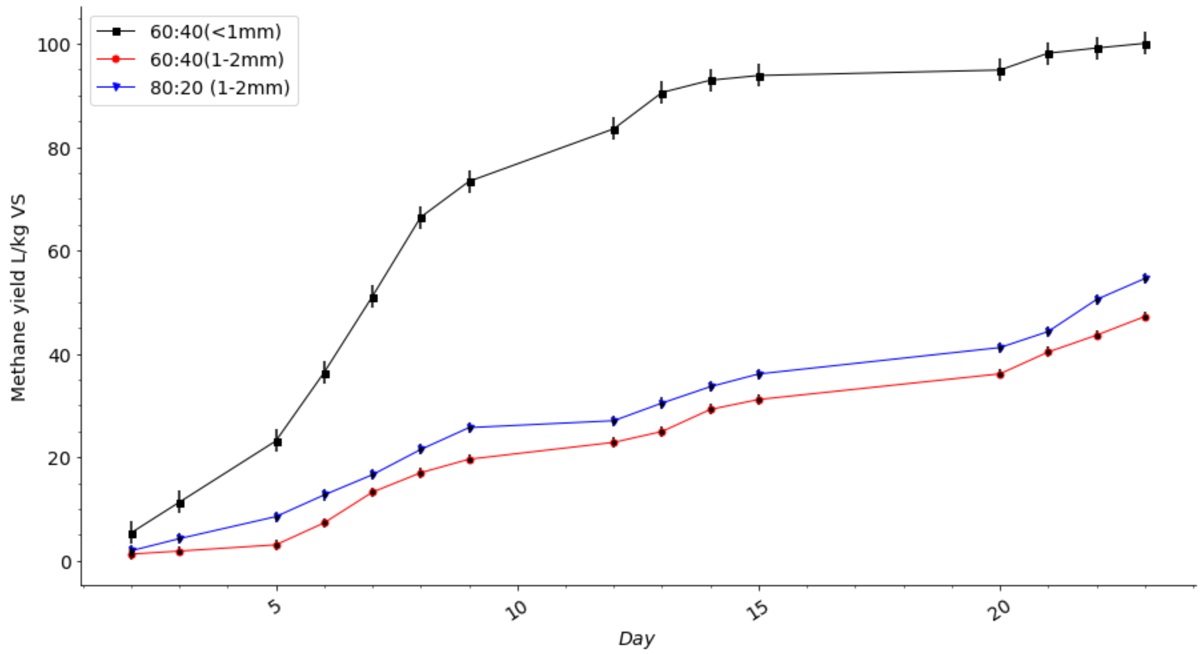


Figure 5.4 - 2: Cumulative daily methane yield for Buffered reactors (L/kg VS)

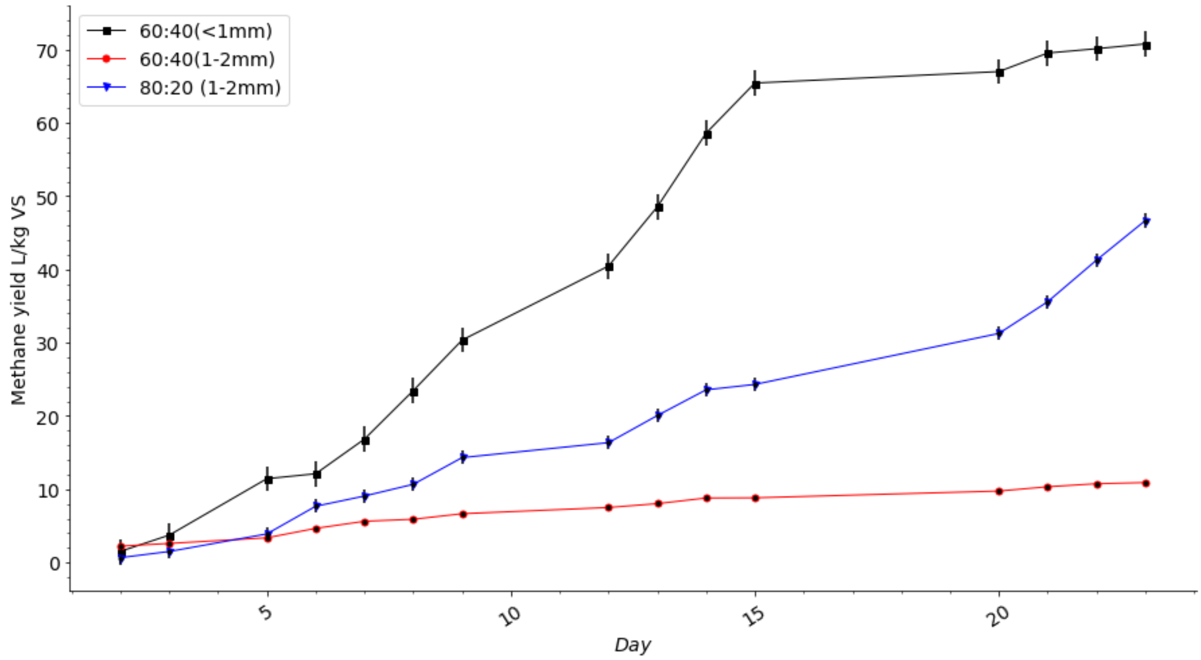


Figure 5.4 - 3: Cumulative daily methane yield for unbuffered reactors (L/kg VS)

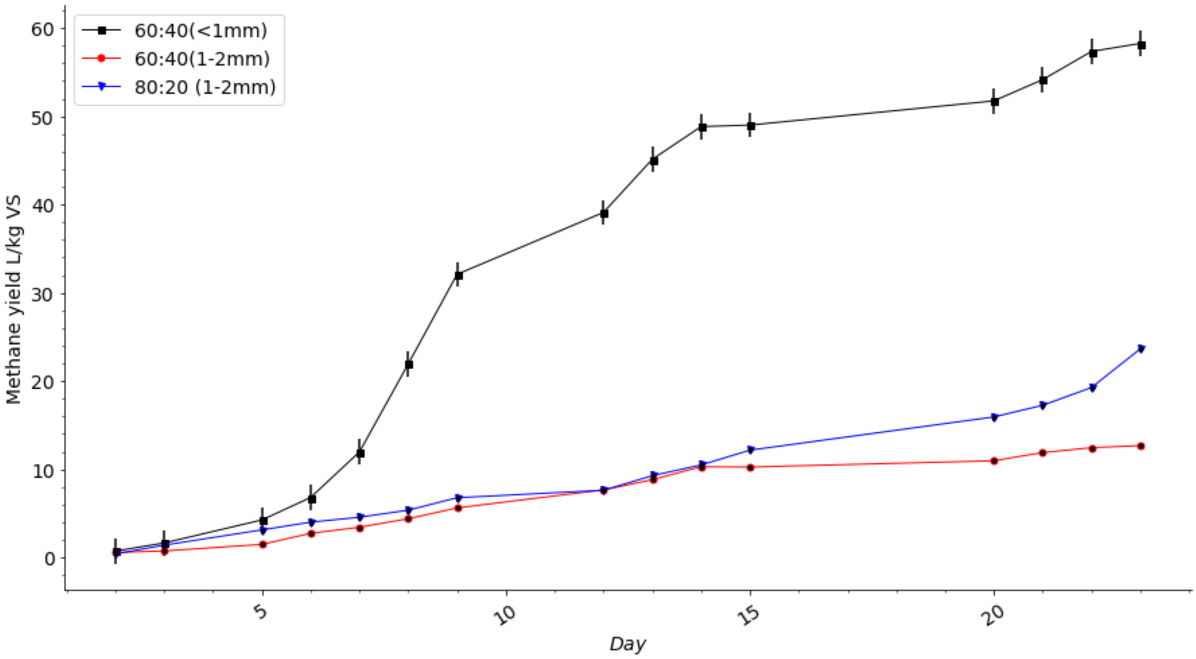


Figure 5.4 - 4: Cumulative daily methane yield for adjusted pH reactors (L/kg VS)

5.4.5 Anaerobic digestion of cattle manure

Figure 5.4-5 shows a graph for the mono-digestion of cattle manure. For every run described above, a triplicate reactor of CD was included. No pre-treatment was done to the CD. Figure 5.4-5 shows the three overlapping methane graphs which is indicative of a good consistency for the runs and the equipment setup. In terms of overall yield, the average methane generation for CD was 36.5 L/kg VS. From the interpretation of the plot, a higher total methane potential is expected on a longer period. CD generated a higher methane yield than co-digestion ratios at low pH which was linked to F&V waste pre-treatment.

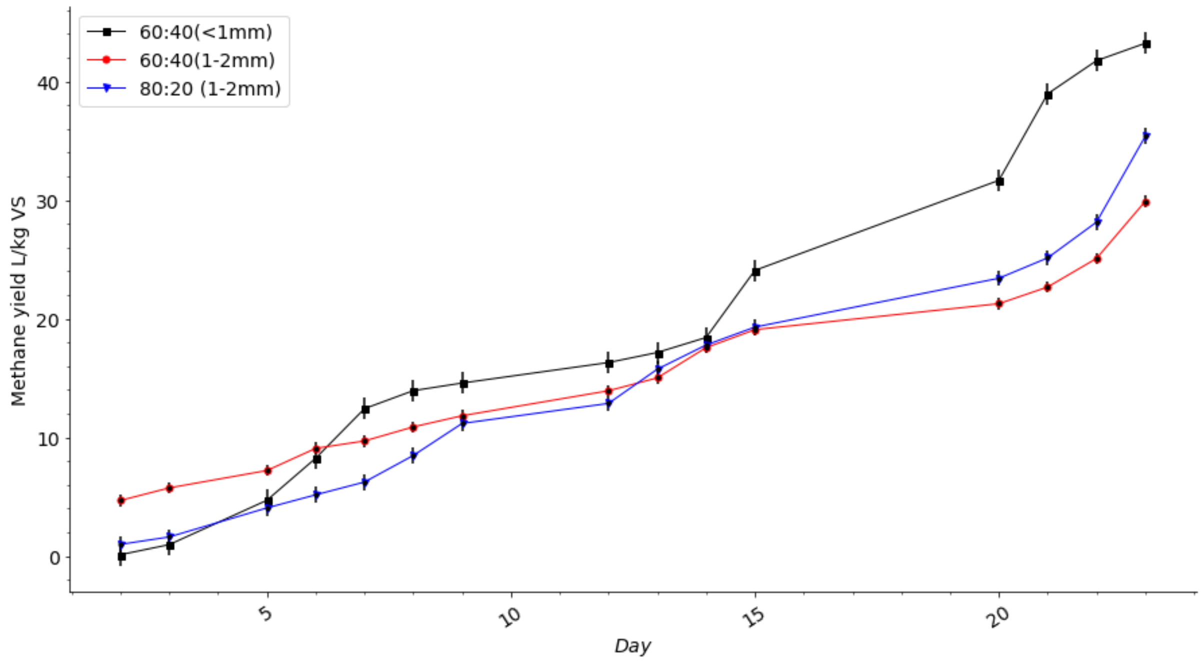


Figure 5.4 - 5: Cumulative daily methane yield for CD (L/kg VS)

5.4.6 Headspace methane analysis

The bioreactors were modified to allow gas readings from the headspace. However, due to the small size of the headspace, only a limited amount could be extracted at a time. A Gas Chromatogram (GC) was used to observe methane content in the headspace. This was performed in the early days of the experiment. At the end of the experiment, a gas analyser was connected to the gas tube to measure methane content. The data to be discussed relates to the 1-2mm runs only. Following the assumption of CH_4-CO_2 totality of biogas, a GC method was established to identify the content of each of the gases. The highest methane purity was observed in the adjusted pH runs at 79.9% and 79.2% for the 80:20 and 60:40 respectively. Methane content in the blanks (CD only) bioreactors followed with a value of 79.8% and 67.7% for the 80:20 and 60:40 respectively. The buffered runs had a methane constitution of 48% and 67% for the 80:20 and 60:40 respectively. During this period, there was little to no volume expelled by the 80:20 run. Methane release became much more prominent on the second half of the observation period. This is further elaborated in the earlier analysis of methane yield graphs (Figure 5.4-2 to 5.4-5). This behavioural response was attributed to the slow digestion due to the pre-treatment used. The 60:40 was already in its advanced methane generation stage. Lastly unbuffered runs had a methane content of 50% for the 80:20 and 75% for the 60:40 run. The results at the end of the experiment showed a different picture to the first part. The volume expelled for 60:40 was at its lowest whilst the 80:20 was yet in the advanced stages of methane generation. The headspace analysis of the buffered bioreactors for the 80:20 runs read a methane content of 85%. The highest daily methane volume was expelled during this period. CO content was 2ppm. The unbuffered 80:20 bioreactors gave an average of 80% with a CO content of 15ppm. Adjusted pH for this ratio recorded a methane content of 95% and a CO content of 21ppm. The blanks gave an average of 90% methane with CO content of 26ppm. Compared to the 80:20 setup, the 60:40 run at the end of the 25 days was almost complete and the gas readings were as follows: Buffered reactors had methane purity of 77% and CO was undetected. Non buffered reactors had a methane composition of 76% was measured with a CO composition of 4ppm. Adjusted pH had methane composition of 85% and a high CO constitution of 100ppm. The blanks had a CO content of 10ppm and a methane content of 78%. From the above observation, despite the low methane volume, adjusted pH runs had the highest methane purity compared to the rest.

5.4.7 Elemental analysis of the digested material

Elemental changes to the digested mass were analysed at the end of the digestion period. This information is useful in determining further treatment of the digested material. Digestate has great usage as fertiliser due to the high presence of soil enriching nutrients. It can also be re-fed as nutrient fuel into the digester. Digestate ultimate analysis is presented in Table 5.4-2 below. The initial amount of C for the substrates was 40% and 34% for the F&V and CD respectively. For the well digested 60:40 (<1mm) substrate, the carbon content reduced to 26.4% for the buffered reactors, 31% for the unbuffered reactors and 25% for adjusted pH. A similar observation was reported in Tawona (2015). The results for the 1-2mm showed a higher C composition. The 80:20 run shows C content of the range 30-40%, whilst the 60:40 (1-2mm) is in the range 39-41% excluding the CD. Several runs contained a C content above 40%. CD runs were digested well and the C content of the digestate was in the range 25-30%. Nitrogen losses were observed for all runs except for the non-buffered 60-40 run. The highest losses were observed for the non-buffered and adjusted pH for the 80:20 and the 60:40 (1-2mm) runs. The S content for the substrates was 0.48% and 0.6% for F&V and CD respectively. This value increased to over 1% in all digestates. S is one of the good nutrients for the soil. Changes to the C/N ratio were also noted and are presented in Table 5.4-2. This ratio was in the range 10-11 for the 60:40 (<1mm), which indicates a high loss of C during digestion. The rest of the experiments shows a much less N content causing the C/N ratios to skyrocket in some cases.

Table 5.4 - 2: Ultimate analysis of Digestate

	Carbon	Hydrogen	Nitrogen	Sulphur	C/N
			60:40 (<1mm)		
Buffer	26.35	3.25	2.23	1.34	11.82
No Buffer	31.77	4.03	2.81	1.52	11.31
Adjusted pH	24.97	3.17	2.26	1.21	11.05
Cow Dung	25.79	3.27	2.43	1.29	10.61
			80:20 (1-2mm)		
Buffer	40.8	5.39	2.24	1.44	18.21
No Buffer	30.70	4.60	0.10	1.02	307
Adjusted pH	39.26	5.30	0.85	1.21	46.19
Cow Dung	29.18	4.97	0.34	1.02	85.82
			60:40 (1-2mm)		
Buffer	39.94	5.21	1.46	1.23	27.36
No Buffer	40.46	5.62	0.25	1.16	161
Adjusted pH	41.13	5.44	0.79	1.29	52.06
Cow Dung	30.60	5.02	1.21	1.15	25.29

5.4.8 Volatile Solids for the digested material

The VS for the digestate was conducted to further elaborate on the extent of digestion of the material. The 60:40 (<1mm) had a VS loss of over 30% for each of the reactions conducted with a final VS of 59% for Buffered reactors, 52.6% for the non-buffered reactors, 54% and 48% for the adjusted pH reactors and CD respectively (Table 5.4-3). This was characteristic of a well digested sludge (WDNR, 1992). The rest of the digestate was not well digestated as very high VS content was found at the end of the experiment. With the exception for CD, the 1-2 mm substrates had a final sludge VS over 70%.

Table 5.4 - 3: Volatile Solids content of the digestate

Buffer	No Buffer	Adjusted pH	Cow Dung
	60:40 (<1mm)		
59	53	54	48
	80:20 (1-2mm)		
76	79	74	51
	60:40 (1-2mm)		
75	79	77	49

5.4.9 Comparing BMP experimental results with theoretical BMP

Elemental and nutrient analysis were used to estimate methane yield as described in **Chapter 2** of this report. This section compares theoretical methane yield and the bench-scale results from this work. It was expected that the theoretical methane yield would be higher than the actual methane generated. Theoretical calculation assumes complete degradation of organic component of the substrate. This is impractical as some of the waste is diverted to support microbial activity during hydrolysis, not all volatile solids are converted and optimality in resource allocation is not guaranteed. Since the 1-2mm substrates were not fully digested, only the <1mm 60:40 was compared in the theoretical plot. The digestion of 60:40 compared well with elemental (ultimate) methane estimation and was only 20% short of the calculated value (Figure 5.4-6). However, it was 49% short of the theoretical methane yield calculated from the nutrient analysis of the waste which estimated close to 199 L/kg VS.

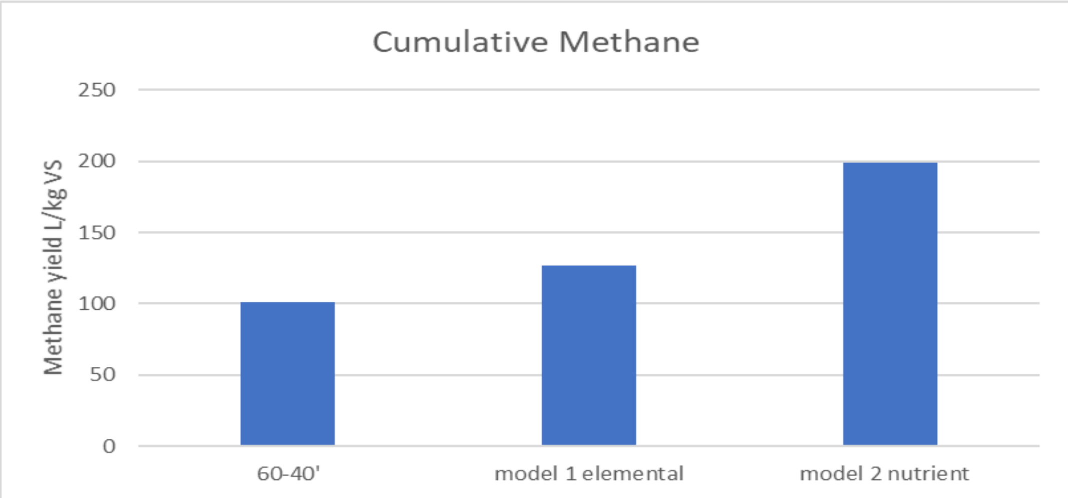


Figure 5.4 - 6: Comparing actual methane yield with theoretical yield

5.4.10 Comparative Analysis with previous studies

The cumulative methane yield for the 60:40 (<1mm) was compared to the yield obtained by Tawona (2015). Tawona (2015) co-digested F&V waste with CD from a similar equipment setup under buffered conditions. However, a higher loading rate of 1.5g/ml VS was used as opposed to the 1.4 g/ml VS used in this work. Figure 5.4-7 compares the daily cumulative methane yields for both sturdies over a 25-day period. Due to the higher loading rate, the methane generation rate was higher for Tawona (2015) compared to this work. For the 25-day period, the total methane generated by Tawona (2015) was 106 L/kg VS in comparison to 101 L/kg VS produced in the current experiment. In assessing the rate of increase, it was notable that the run by Tawona (2015) reached the highest daily rate earlier than the experimental data from this work. Thus, reaching the highest methane potential for the substrates in a shorter period. This is demonstrated by the great disparity in the trend line at the beginning of the run that is rapidly closing only after 2 weeks in Figure 5.4-7 and the daily methane yield shown in Figure 5.4-8. The daily methane yield shows a peak in methane generation only after 2 days, reaching the highest rate in 5 days for Tawona (2015). This was followed by a gradual decline with occasional picking until day 25. The weekly rates presented in Table 5.4-4 shows that within the first week of data collection over

70 L/kg VS was measured which was 66% of the total methane generated in 25 days for Tawona (2015). The current experiment observed its highest peak in 8 days after a gradual increase from a peak at day 5. The highest daily methane generated was 15.3 L/kg VS at day 8 compared to 19.2 L/kg VS measured by Tawona (2015) three days earlier. Within the first week over 50% of cumulative methane was produced as compared to 66% from Tawona (2015). After two weeks, 93% of the cumulative methane was released compared to 92% for Tawona (2015). Average daily methane generation after 14 days was 1.4 L/kg VS for the current work and 1.7 L/kg VS for Tawona (2015). 98% and 99% of methane was generated by the third week of the experiment for the current experiment and Tawona (2015) respectively. This result is very characteristic of the digestion of F&V substrates which are known to undergo quick hydrolysis. Buffering ensured that the accumulation of acid did not cause a rapid drop in pH, inhibiting the activity of methanogens. Thus, one can conclude that under buffered conditions, the AD of F&V in co-digestion with CD can yield over 90% of methane in just two weeks. A higher loading rate can be trusted to provide a higher total yield and as well a faster conversion in the first seven days of running a digester.

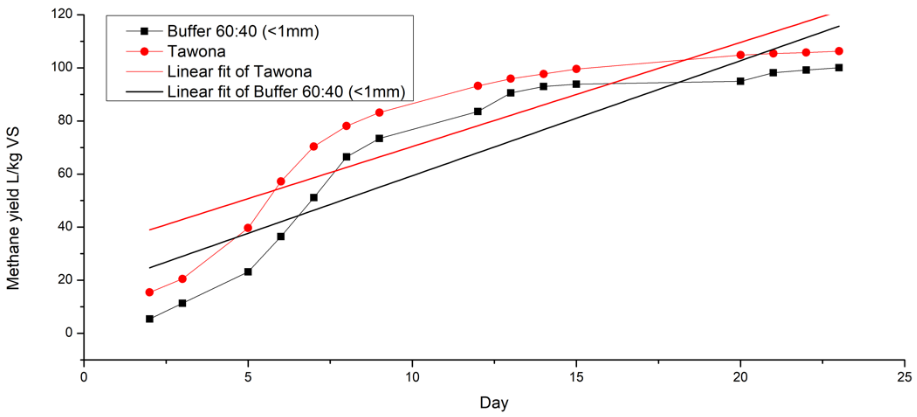


Figure 5.4 - 7: Comparing methane yield with (Tawona, 2015)

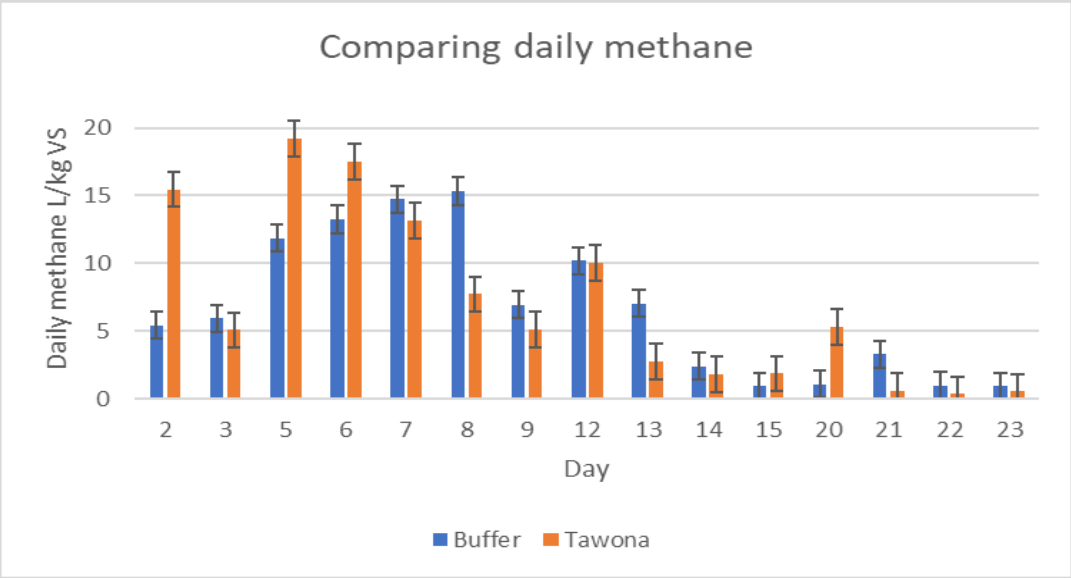


Figure 5.4 - 8: Comparing daily methane yield with (Tawona, 2015)

Table 5.4 - 4: Comparing cumulative weekly methane yield

Week	**Buffer** L/kg VS	**% gain**	**Tawona** L/kg VS	**% gain**
1	52	51	54	66
2	79	93	74	92
3	79	98	77	99

5.4.11 Repeatability of the BMP experiment

Repeatability of the reactions performed in this work was done in accordance with ISO 5725. Repeatability is calculated as $1.6S_r$, where S_r is the standard deviation of the samples in the series (Hansen *et al.*, 2004). This value provides an uncertainty in methane yield of repeated measurements of the same variable investigated. It provides "a 95% envelope for absolute difference between two single measurements of the same sample" (Hansen *et al.*, 2004). Three runs of a triplicate set of CD was used to measure the repeatability in this work. The series had a standard deviation of 6.69 and yielded a repeatability for the experiment of 10.71 L/kg VS.

5.4.12 Feasibility analysis and recommendations for upscaling

The underlining concern surrounding every bench-scale research is the feasibility of its upscale based on real life compositional data. Setting up an AD plant requires a constant organic feedstock supply and a consistent methane generation which is affected by the composition of the substrates. Some of the main parameters include plant capacity, total feedstock properties, retention time as well as the operating temperature (IRENA, 2016). These parameters are detrimental in determining the necessary technological approach and equipment sizing for all the processes concerned from pre-treatment to gas clean-up. The problem of waste management is the limiting factor to establishing the viability of the technology at a larger scale. Based on the data in **Chapter 3**, as per the lack of separation of organic waste by the municipal and the condition of black plastic packaged conglomerated waste, it remains a far-fetched idea to size a biogas plant reliant on the unknown data. Thus, several recommendations are stated as per the outcome of this research for Durban Solid Waste (DSW) and these include:

- Performing a cost analysis on the possibility of separate collection of organic waste to a separate facility

- Prohibition of large food-chains from landfilling organic waste but rather diverting as much as possible and if a must dispose, it should be sent to an organic waste facility. At these facility, mechanical and biological treatments can be undertaken on the waste both for valorisation and stabilisation purposes.

- Education of the public in both rural and urban settings on the importance of handling organic waste as a valuable resource

- Performing an assessment on the usability of micro-digesters in remote areas for methane generation which can give rise to opportunities for the youth in the area

Micro digesters have drawn much attention in the recent years. These are small scale household reactors which are not as sophisticated to build and operate as the industrial digesters. In India and China, millions of small-scale digesters are used in rural settings for cooking purposes (Rajendran, Aslanzadeh and Taherzadeh, 2012). The commonly used micro-digesters are fixed dome, floating drum and balloon/bag digesters (IRENA, 2016). The maximum methane yield recorded in this report was 101 L/kg VS from the buffered reactors. Using the Clairwood market as a case study and estimating the availability of organic feed per annum at 768 tonnes (30% of 2560 tonnes), the total volume of methane generated is estimated at 46 540 m^3 per annum, which translates to 128 m^3 per day (365-day calendar). The calculation was based on a 60:40 (<1mm) co digestion ratio (CD = 1152 tonnes) and a combined VS of 23.84%. IRENA (2016) estimated that 1 m^3 of methane equals 34 MJ of energy dissipated. Thus, 128 m^3 would correspond to a total production of 4 352 MJ of energy per day. Using equation 5.4-1 below, the volume of reactor required would be 2.91 m^3. For a fixed dome plant with a hemisphere design, this volume translates to a reactor diameter of 2.4m and a total plant volume of 3.62 m^3 (Table A-6).

$$G = \frac{Y \times V_d \times S}{1000} \qquad \text{Equation 5.4-1}$$

Where:

G(m^3/day) = biogas production

Y = yield factor

S(kg/m^3) = initial concentration of VS

V_d = digester volume

5.4.13 Section summary

Anaerobic co-digestion of F&V waste and CD was performed under low solids content. The parameters investigated were pH, co-digestion ratio as well as the amount of pre-treatment. Pre-treatment proved to be the most determining factor to the digestibility of the organic substrates in comparison to pH and co-digestion ratio. A smaller sized substrate can effectively digest and in a short period of time compared to a larger particle sized substrate, regardless of the pH used nor the co-digestion ratio. Optimality in terms of pH was obtained at a high pH between 7.5 and 8.5. A buffered reactor with a pH in that range provided the highest methane compared to a buffered reactor with a pH in the 6.5 to 6.8 range. Even unbuffered reactors, with a co-digestion ratio of 80:20 for CD to F&V provided a higher yield compared to low pH buffered reactors. Furthermore, a co-digestion ratio of 80:20 provided better results than a co-digestion ratio of 60:40, which experienced inhibition under unbuffered conditions. A 60:40 set with a particle size less than 1mm was well digested and the resulting VS of its digestate were between 53 and 59% compared to over 70% for the inefficiently digested 60:40 and 80:20 substrates with particle size in the range 1-2mm. Well digestated substrates were characterised by a low carbon (C) content and a high nitrogen (N) and sulphur (S) contents.

5.5 Characterisation of Digestate and Feasibility Analysis

This section further analyses the digestate qualities and demonstrates its usability as a fertiliser and in pyrolysis.

5.5.1 FTIR analysis of digestate

The 60:40 (<1mm) digestate was analysed under FTIR to assess the changes to the functional groups after the digestion process. The graphs show overlapping bands as expected, indicating the presence of same functional groups in raw and digested states. However, the level of transmittance varies in accordance with the utilisation of nutrients during the digestion. Figure 5.5-1 shows the comparison of each of the four digestates discussed in **Chapter 4**. In assessing the no buffer digestate, a decline in transmittance for most peaks was observed. The most characteristic was the broad peak vibrating at 3290 cm^{-1} and a sharp, pointed peak at 1032 cm^{-1}. The peak at 3290 cm^{-1} is associated with an O-H absorption for phenols, alcohols or carboxylic groups and C-H for alkyl stretching groups. Spectral absorption at 1032 cm^{-1} is a C-O group, representative of the presence of polysaccharides (Provenzano *et al.*, 2011). Adjusted pH digestates shows characteristic peaks at 3290 cm^{-1} and 1032 cm^{-1} which have significantly lower transmittance than the starting material. These however have a higher transmittance than No buffer digestates. Both Adjusted pH and Buffer digestates spectra plot maps CD transmittance for the most part. Much of the spectra for the cow dung digestate (Blank) was the same as that of the starting CD substrate except from 611 – 1634 cm^{-1}. Low transmittances were observed at the characteristic absorbances 2918 and 2880 cm^{-1} indicative of fatty acids structures.

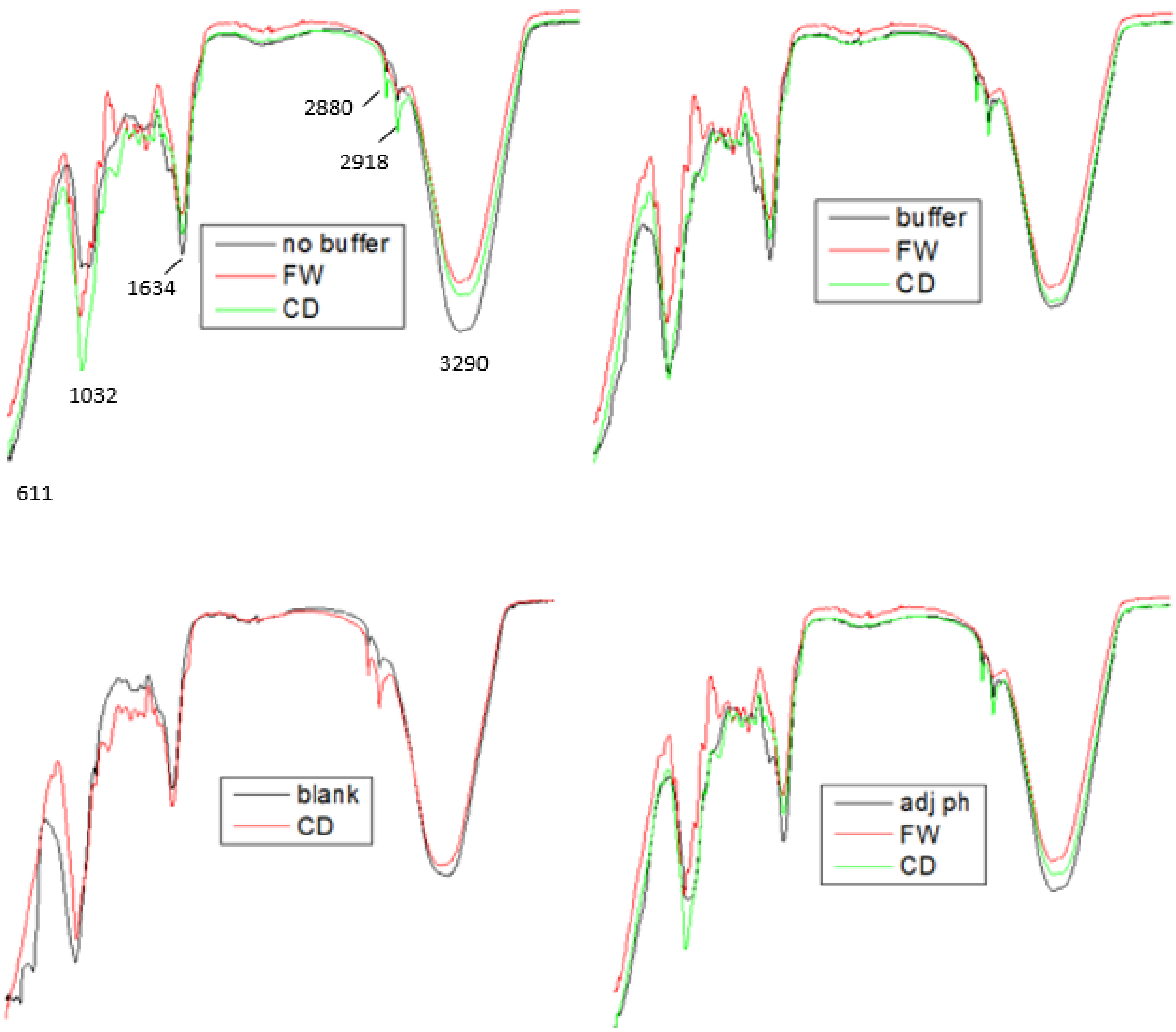


Figure 5.5 - 1: Functional group analysis of digestate

5.5.2 Thermal degradation of the digestate

Thermal characterisation is important in establishing a potential relationship between chemical components and the thermo-degradation process (Wei, Hong and Ji, 2018). The thermal characterisation of digestate is depicted in Figure 5.5-2. In the plot, sample weight loss as well as the differential of weight with respect to temperature is presented. This is important in determining the chemical composition of substances. The thermal degradation of the digestates was very similar, almost overlapping. This was because of the same starting substrates. The total weight loss varied amongst the digestates. This corresponded and thus associated with the available volatile solids (VS) after digestion. The highest weight loss was observed on the buffered digestates. A 46% loss in weight was measured, which corresponded to the VS of 59%, the highest amongst the digestates. CD had the least loss in weight of 36%, corresponding to the lowest VS of 48%. Non-buffered and adjusted pH digestates incurred weight losses of 36% and 37%. The remaining char was 54%, 64%, 63% and 68% for buffer, non-buffered, adjusted pH and cow dung digestates respectively. There were two clearly defined regions of weight loss that were observed from the differential plot (DTG) of the digestates. The first region accounted for about 5% of the weight loss for all the digestates. This peak was attributed to the removal of moisture from the samples. The second stage was simultaneous with the lengthy and steady weight loss observed between 200ºC and 350ºC. A weight loss of about 30-35% was measured. The well-established peak corresponded to the degradation of cellulose which degrade in the area defined between temperatures of 250 ºC and 380 ºC (Díez *et al.*, 2020). The much lesser defined peaks that overlaps the cellulose peak are the hemicellulose and lignin peaks (Diez *et al*., 2020). These peaks are more defined in other digestates than others. The degradation of hemicellulose occurs between 200 ºC and 300 ºC and is indicated by a small peak that forms on the left shoulder of the cellulose peak, whilst the degradation of lignin is more complex and occurs over a wide range of temperature from 200 ºC to 1000 ºC (Diez *et al*., 2020). A small peak emanating on the right shoulder of the cellulose peak at around 400 ºC was identified as a lignin peak.

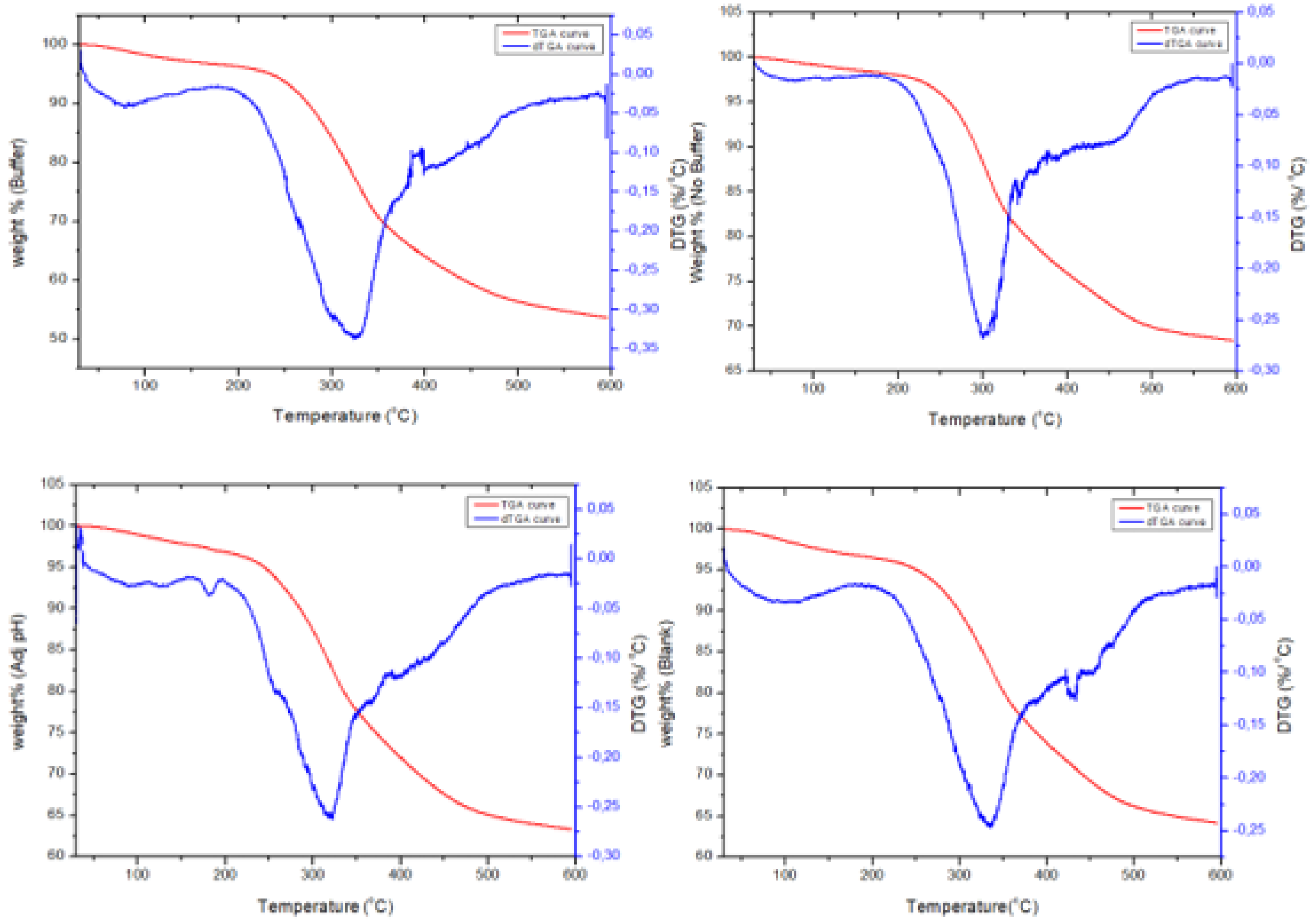


Figure 5.5 - 2: Thermal degradation of the digestates

5.5.3 Mass balance

The type of digester used in this research was a low-solids, mesophilic batch digester. The fixed feedstock was kept for a residence time of 25 days at a constant temperature of 36^{o}C. A fixed substrate feed of 12.2g was digested. The total volatile solids (VS) of the substrate feed at an organic loading rate of 1.4g/100ml VS was 8.12g VS. A ratio of 60:40 in terms of VS was used for CD and F&V waste respectively. The dry mass analysis of the digestate is reported in Table 5.5-1. The Buffered run eliminated 43% of the initial substrate mass leaving behind a digestate with a VS of 59%. The non-buffered and adjusted pH digesters retained 76% and 75% of the initial mass feed with a VS content of 53% and 54% respectively. Cow Dung retained a VS content of 48% to the 79% remaining dry matter.

Table 5.5 - 1: Mass analysis of digestate

Buffer	No Buffer	Adjusted pH	Cow Dung only
	Total dry mass/starting mass %		
57	76	75	79
	Volatile Solids %		
59	53	54	48

5.5.4 Digestate usability as a fertiliser

The use of digestate as a fertiliser is the best sustainable way to recycling organic matter and important nutrients back into the soil (Taylor *et al.*, 2008). The usability of digestate as a fertiliser relies not only on the nutrient capacity but also on the amount of hazardous matter of chemical and physical nature that can cause damage to the soil and the environment (Al Seadi *et al.*, 2013). Thus, a high quality digestate is defined by a high organic matter content, at a good pH, that is rich in nutrients and free from pathogens such as stones and plastics. To produce a high quality digestate, a high-quality feedstock must be used (Al Seadi *et al.,* 2013). Much of the chemical and physical pathogens can be removed by physical means and thus a proper collection, storage and particularly physical pre-treatment of feedstock must be ensured to improve the quality of feedstock and to a greater extent the digestate. Digestate usage as a fertiliser can improve the plant availability of nutrients. According to Moller and Muller (2012), plant available N and uptake efficiency of slurry N is higher for digested material. It is however important to apply the fertilizer as soon as possible after the digestion process. This is because it has been proven that significant N losses results from storage of the fertiliser and when it is dried before application (Moller and Muller, 2012). The same was said for other essential soil nutrients such as S and P. The best results in terms of nutrient uptake efficiency were observed for application in spring compared to autumn (Moller and Muller, 2012). Digestates can be further treated before application. Common treatments include solid-liquid separation, pelletizing as well as composting (Al Seadi et al., 2013; Moller and Muller, 2012), however, as already stated, these may not serve as best solutions for usage as fertiliser. Composting is best used for poor digestates that cannot be used as fertilisers and can rather be used as landfill covers or in other applications. Section 5.5 presents the chemical analysis of the digestate obtained in this work. This information in important in determining the usability of the digestate as a fertiliser.

5.5.5 Chemical analysis of digestate

The chemical analysis of digestate from this work is presented in Table 5.5-2. Post-digestion treatment of digestate was conducted and the solids and liquids were separated and dried before analysis. CHNS analysis is important in determining the nutrient capability of the digestates. Important elements are further discussed in this section in relation to the post digestion usability of the digestates as well as pollution potential. The total N was measured between 2.2 and 2.4%. This is typical due the nutrient losses during the separation and subsequent drying of the solids, Moller and Muller (2012) reports that due to such losses, nitrogen content is expected to be in the range of 2-3% in the solid sample and only as high as 7% in the liquid digestate. Thus, both the liquid and solid digestates finds great usage as soil improvers. Another major soil nutrient, S was in the range of 1.21 to 1,52% in the digestates. The least amount of S was found in adjusted pH digesters, which were operated at a low pH of 6.5. This low content was attributed to the losses owing to the conversion of S to H_2S which is highest at low pH values (Yang *et al.*, 2015). The highest S content was in buffered and non-buffered reactors which operated at a high pH value. Carbon to nitrogen ratio in the soil defines the efficiency in nutrient availability of the crops from the soil. The optimal C:N ratio for crops is 24:1 whereas higher ratios lead to slow microbial decomposition whilst a low C leads to a faster decomposition (USDA, 2011). The ratio of C to N was in the range of 10.61 and 11.82 in the digestates. This was attributed to a high degradable substrates feed (Moller and Muller, 2012). The biological and chemical degradability of the digestate was measured. The degradability of digestates indicates the possible extent in the pollution potential as the materials continues to degrade in an uncontrolled environment. The Biological Oxygen Demand (BOD) was conducted over a 7-day period and the results are presented in Table 5.5-2. The highest biodegradability was 274 mg/l for a low pH digestate (adjusted pH). This value was significantly higher than for the other digestates. It was seconded by the buffered digestates with 89.1 mg/l, a degradability that is 67% lower than that of adjusted pH digestate. The lowest biodegradability was measured for the unbuffered digestate with a value of 57 mg/l. Cow Dung (CD) digestate had a BOD_7 value of 64.1 mg/l. A similar trend was observed during the measurement of the Chemical Oxygen Demand (COD). The COD for adjusted pH (pH<7) was the highest with 4456 mg/l. Both measurements of biodegradability show that under adjusted pH, the least amount of degradation of the organic component was done. The least amount of COD was measured for the buffered digestates. Lastly, CD had a COD of about 928 mg/l.

Table 5.5 - 2: Chemical analysis of digestate

Buffer	**No Buffer**	**Adjusted pH**	**Cow Dung only (Blank)**
		BOD7 mg/l	
89.1	57.0	274	64.1
		COD mg/l	
835.52	2351.82	4456.08	928.35
		Digestate pH	
7.41	9.85	6.72	10.97
		Total Nitrogen %	
2.23	2.81	2.26	2.43
		Sulphur content %	
1.34	1.52	1.21	1.29
		C/N ratio	
11.82	11.31	11.05	10.61

The estimation of other components contained in the sample was done using an Energy Dispersion X-ray (EDX) machine that is equipped with a Scanning Electron Microscope (SEM). This machine was used mainly to detect the various metals that are contained in the digestate. a precise amount could not be measured. A large amount of undigested material was composed of soil particles. This is noted in Table 5.5-3 by the large amount of Silicon (Si) in each of the digestate. The soil must have been introduced to the substrates during the collection of CD from the cattle post. Apart from potentiality to destroy AD equipment during operation, this component will have no negative impact to the soil upon recycling. The most common metals among the adjusted pH and Cow Dung digestates were Strontium (Sr) and Niobium (Nb). The important soil improving components (N, P, K) were not identified in most of the digestates. Potassium (K) was found in a buffered digestate at 3% and only 1% in an unbuffered digestate. Traces of phosphorus (P) were identified under unbuffered digestate as well as nitrogen at 1% and 4% each, respectably. A more vigorous equipment is required to take an accurate measure of the metal components and their weights.

Table 5.5 - 3: Metal analysis of digestate

Metal	Buffer %	No Buffer %	Adjusted pH %	Cow Dung only %
O	36	15	43	43
Si	25	11	35	35
Nb	1	4	10	10
Sr	2		11	11
Al			1	<1
Br	6			
N		4		
P		1		
K	3	1		
C	36	63		
Na	<1	2		

5.5.6 Digestate usability as feedstock for pyrolysis

Pyrolysis of digestate was performed at two temperatures, 300°C and 500°C. These temperatures were selected in order to assess the effect of temperature on the distribution of the pyrolysis products, oil and char. The results are presented in Table 5.5-4 and analysed in Figure 5.5-3. There was a large amount of condensate obtained during the pyrolysis of digestates. The generation of bio-oil was significantly higher than biochar and biogas at both temperatures. At 300 °C, bio-oil accounted for 78% of the product stream. The second most abundant product at this temperature was syngas/bio-gas with 17.6% and lastly biochar with 4.4%. At 500 °C the product distribution was slightly different with biochar being slightly higher than biogas at a comparable value of 18.5% and 18.1% respectively. A characterisation of both the bio-oil and biochar is presented in this chapter. Syngas was not in the scope of this research and was thus not analysed.

The effect of temperature on the product distribution is shared in Figure 5.5-4. Biochar production increased with an increase in temperature. The wt.% of the biochar at 300 °C was 4.4 wt.% and increased significantly to 18.5% when the temperature was increased to 500 °C. Therefore, for pyrolysis with the aim of producing biochar, a higher temperature is recommended. The optimal temperature is determined based on the desired output. However, at both temperatures the amount of bio-oil generated was very high, ranging from 63% to 78% at 500 °C and 300 °C respectively. There was a 15 wt.% drop in bio-oil generation when the final temperature was increased from 300°C to 500°C. Thus, unlike biochar, the generation of bio-oil decreased with temperature. The relative abundance of the product thereof makes it a very important product of pyrolysis, requiring proper handling and end usage regardless of whether it is the desired product or not.

Table 5.5 - 4: Pyrolysis of digestate

Temperature °C	**Starting mass g**	**Biochar** wt.%	**Bio-oil** wt.%	**Syngas** wt.%
300	46.5	4.4	78.0	19.8
500	50.4	18.5	63.4	18.1

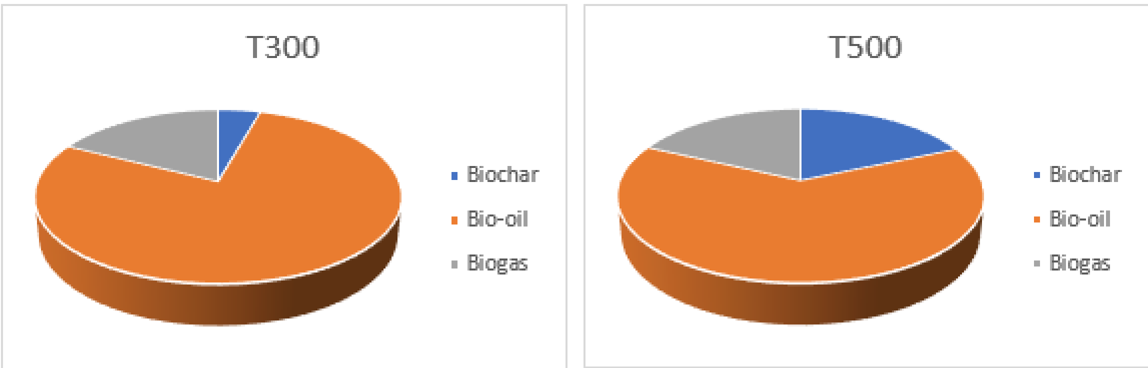


Figure 5.5 - 3: The distribution of pyrolysis products

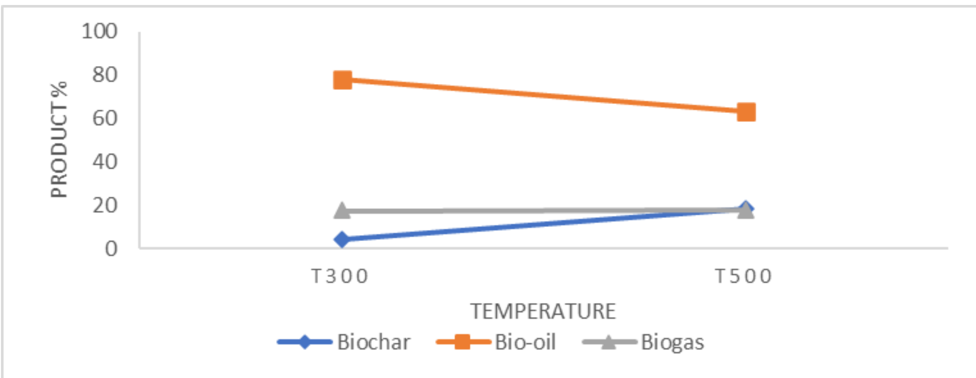


Figure 5.5 - 4: The effect of temperature on the distribution of pyrolysis products

5.5.7 Characterisation of biochar

There are several important usages of biochar. It can be used as an alternative to mineral fertilisers and have been proven to provide comparable results. Biochar can furthermore be used as an alternative to charcoal because of a high heating value contained. Table 5.5-5 provides characterisation information for the biochar. A higher amount of Carbon and Nitrogen was observed at 500ºC compared to when the pyrolysis was conducted at 300ºC. The carbon content increased by over 18% whilst hydrogen observed a 3.83% increase from 1.02% at 300ºC. The same observation was made with the sulphur content of the samples. The sulphur content increased from 0.42% to 1.06%. There was a nitrogen loss, however, as the temperature was increased. At a lower temperature, 1.71% of the biochar constituted nitrogen, whilst at 500ºC, the content reduced by 0.9% to 0.80%. The C/N ratio for both biochar was outside the optimal ratio of 24:1 defined by USDA (2011). At 300ºC, the C/N ratio was about half the optimal value at 12.06:1, whereas, at 500ºC the ratio increased to double the optimal ratio. In comparison with the digestate, the biochar improved the C/N ratio of the digested mass. A nitrogen and sulphur loss was observed after charring, whereas a gain in carbon and hydrogen was observed at 500ºC. In comparison to literature, pyrolysis performed by (Opatokun *et al.*, 2015) on digested food waste at a temperature of 500ºC with a rate of 10ºC/min, results for biochar characterisation measured a C/N ratio of 8.8 with higher values for nitrogen and sulphur compared to the biochar from this work. The higher heating value (HHV; dry basis) of biochar was found to be 10.83 MJ/kg and 2.17 MJ/kg for pyrolysis at 500 ºC and 300 ºC respectively. The heating value for biochar at 300 ºC was lower than that of the digestate. However, the heating value for the biochar at 500 ºC was 63% higher than that of digestate which was calculated to be 4.30%. This result demonstrated the importance of pyrolysis in improving the heating value and thus the usability of digestate in heating applications. The following equation was used to estimate the HHV value (Martín-Lara *et al.*, 2021).

$$HHV\ (MJ/kg) = 0.314 * C + 1.322 * H - 0.12 * O - 0.12 * N + 0.0686 * S - 0.0153 * Z\ (\text{ash fraction}) \qquad \text{Equation 5.5-1}$$

Table 5.5 - 5: Characterisation of biochar

	Carbon	Hydrogen	Nitrogen	Sulphur	Oxygen	C/N	Heating Value MJ/kg
Char T_{300}	20.62	1.02	1.71	0.42	76.23	2.06	1.56
Char T_{500}	38.79	4.85	0.80	1.06	54.5	48.49	11.75
Digestate	27.22	3.43	2.43	1.34	65.58	11.20	4.30
Opakotun et al., 2016	35.3	1.2	4.01	4.39		8.80	

*O content was determined by difference method

A similar break-down of the metal distribution in the biochar compared to the digestate was observed. At 300ºC, a high amount of C, O and Si was observed with small amounts and traces of other components (Table 5.5-6). The same was noted for biochar from a pyrolysis at 500ºC but with the exception to Na which was also available at a high amount. The amount of nitrogen and phosphorus increased when the temperature was raised to 500ºC.

Table 5.5 - 6: Metal analysis of biochar

Metal	T_{300}	T_{500}
O	31	55
Si	17	6
Nb	1	1
Sr	4	<1
Al	1	<1
Br	<1	<1
Te	2	1
N	<1	1
P	<1	2
K	<1	<1
C	42	15
Na	<1	10
Ca	<1	<1
Mg	<1	<1

5.5.8 Characterisation of bio-oil

Bio-oil was characterised using a GCMS. The peak integration information is presented in Figure 5.5-7 and 5.5-8. There was a wider variety of peak distribution at 300 °C in comparison to 500 °C. Peak integration results were categorised as shown in Figure 5.5-9 and Figure 5.5-10. The product was categorised as carboxylic acids, esters, alcohols, amides, amines, phenols and ketones. These groups were generalised and grouped as per the main group. Phenols included guaiacols and syringols. There were three major groups from the characterisation of bio-oil at 300 °C. The product distribution was led by ketones with an abundance of 29.65% and the second and third highest occurring groups were phenols and ketones with 27.44% and 27.01% respectively. The fourth abundant were others. Others referred to compounds that did not fall into the groups stipulated. This category comprised of compounds such as methyl chloride and carbon dioxide and had a proportion of 6.42%. The least occurring group was alcohols with an abundance of 0.58%.

The bio-oil produced at 500 °C comprised majorly of carboxylic acids. The acids had an abundance of 47.61% and 42.38% of this composition was acetic acid. Phenols were the second highest with 17.12% and followed by ketones with 15.02%. At this temperature, no alcohols nor other compounds were identified in the oil. The least occurring group was esters with an abundance of only 2.75%. Amines and amides had a proportion of 10.95% and 6.55% respectively.

Comparison of the effect of temperature on the composition of bio-oil was prepared and the results presented on Figure 5.5-11. The first observation was a momentous decrease in the abundance of the formerly major groups when the temperature was increased. The ketones dropped from 29.65% to 15.02% whilst the phenols dropped by 10.32% to 17.02%. However, carboxylic acids increased in abundance from 27.01% to 47.61% as the composition of acetic acid increased from 16.78% to 42.38%. The relatively high amount of acids was attributed to the highly acidic fruit and vegetables which was in the starting substrate. Amines and amides increased in abundance whilst the esters and alcohols decreased.

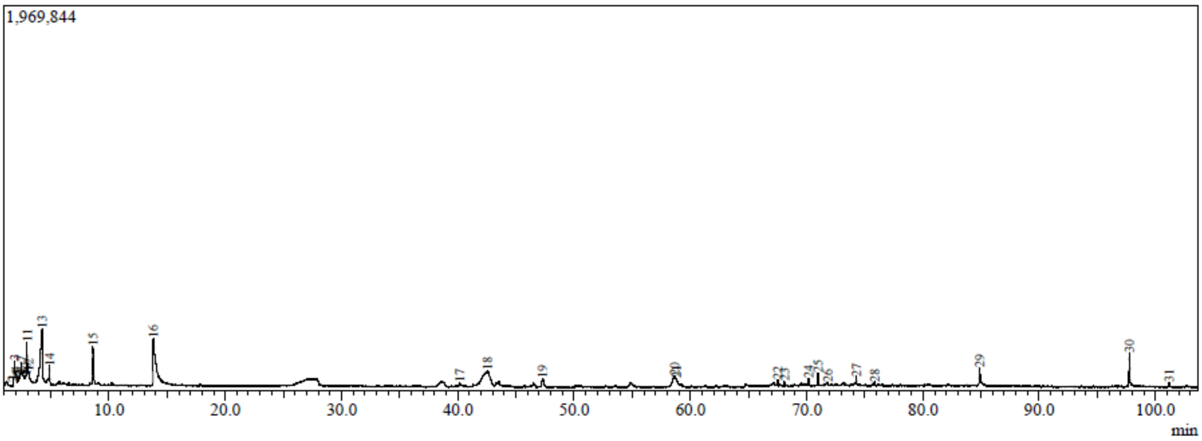


Figure 5.5 - 5: Peak integration graph for bio-oil at 300 ^{o}C

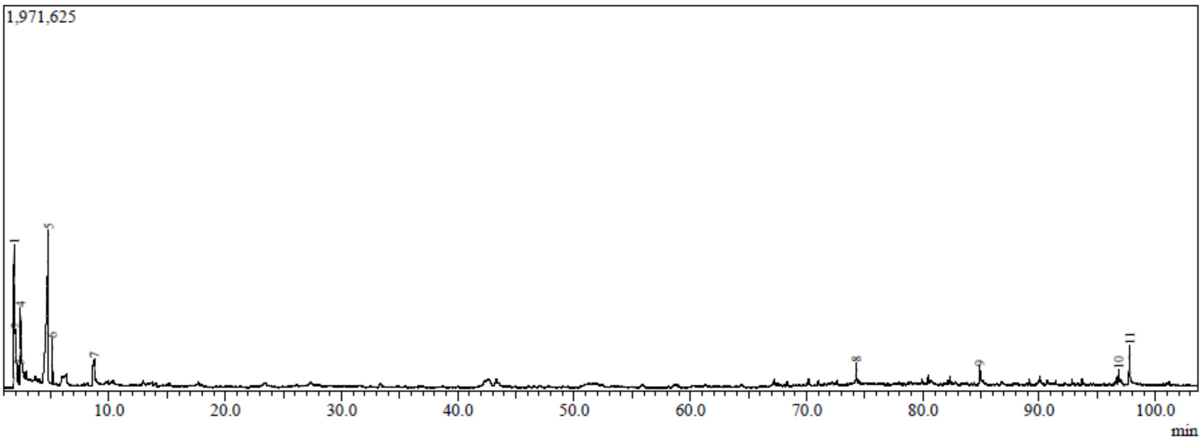


Figure 5.5 - 6: Peak integration graph for bio-oil at 500 ^{o}C

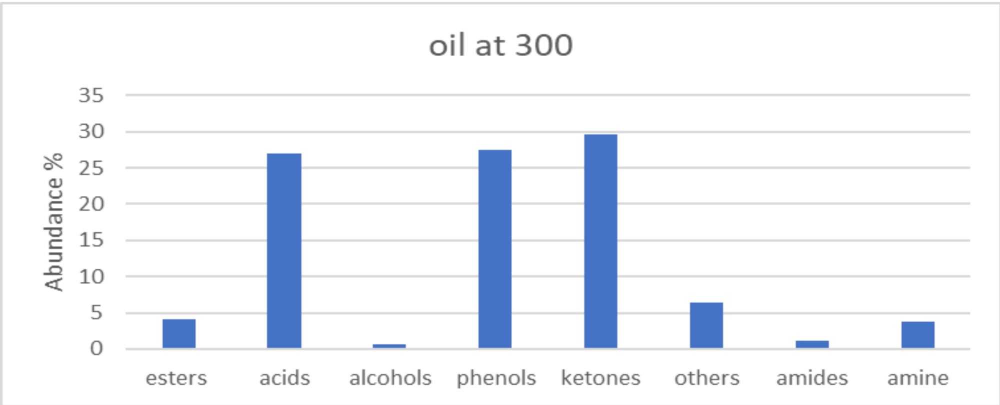


Figure 5.5 - 7: Bio-oil product distribution at 300 ^{o}C

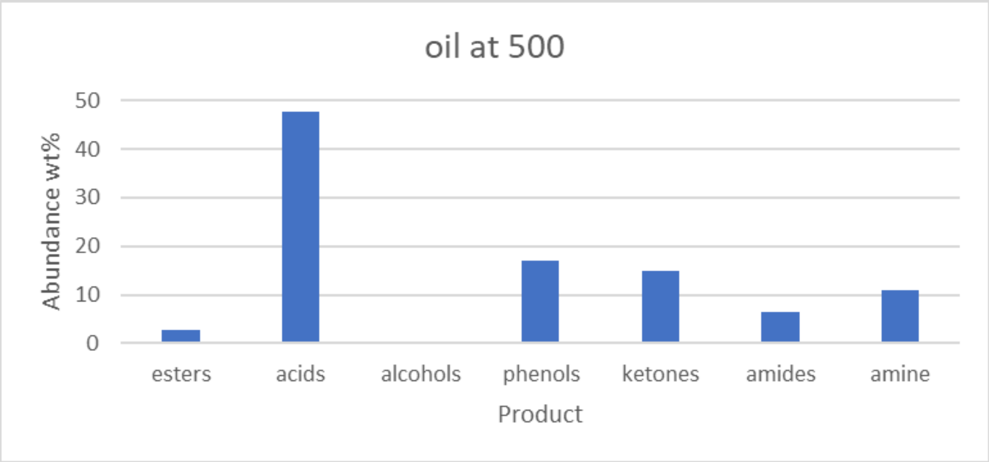


Figure 5.5 - 8: Bio-oil product distribution at 500 °C

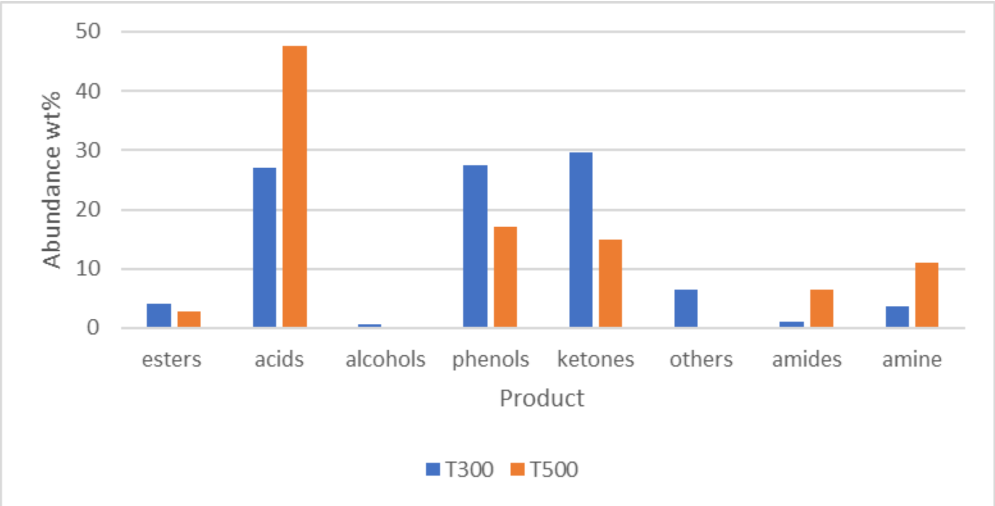


Figure 5.5 - 9: Comparison of bio-oil distribution at 300 °C and 500 °C

The use of these lignocellulosic materials are complex and have very important industrial applications. Carboxylic acids find great usage in the food industry, medicinal fields and as adhesives coatings (Badea and Radu, 2018). Phenols as well finds application in medicinal industries and in household products amongst others (Wade, 2018). Esters as well as alcohols are used in perfume making and as synthetic flavours and in cosmetics (Brown, 2019). Ketones have medicinal usage and can also be used as solvents (Brown, 2019). Amines are used in pharmaceuticals and in the industries as corrosion inhibitors in boilers and water proofing agents in textiles (Peter, 2018). Lastly, amides finds usage as solvents and in fertilisers amongst others (Augustyn, 2019).

5.5.9 Section summary

In this section, post treatment of digestates was done and the valorisation of the digestate investigated. The nutrient analysis for the digestates was compared to literature and reported. The usability of digestates as a soil improver was done. It was found out that there was a rich nutrient presence in the digestates, however, an optimal amount in relation to the soil requirements was not met. A further investigation on the usefulness of digestate as a potential feedstock for pyrolysis provided a nutrient rich biochar and syngas. The amount of biochar was prominent at high temperatures compared to the oil and conversely at low temperatures. In analysis of the pyrolysis products, the biochar showed a great potential in improving the nutrients capacity of the digestates and thus the usability as a soil improver. Furthermore, pyrolysis oil was rich in carboxylic acids and phenols which are very useful industrially. Lastly, it was demonstrated that pyrolysis conducted at 500 ^{o}C improved the heating value of digestates by over 63%.

6. CONCLUSIONS AND RECOMMENDATIONS

This project assessed the feasibility of utilising Anaerobic Digestion technology in the conversion of Municipal Organic Waste to biogas. A literature review of existing data pertaining the characterisation of the waste stream was conducted and it was found out that the data was very general and could not be relied upon when sizing an AD plant. This was mainly due to the lack of separation of waste at source which made it difficult to perform a thorough waste characterisation. Furthermore, the legislation still permits the landfilling of all forms of organic waste. It was found out that stabilising of unsorted waste at landfill can be achieved through Mechanical Biological Treatment (MBT). This technology, however, still works effectively if there is some sort of separation of waste at source.

A case sturdy of the major landfills in Durban was conducted to assess the waste management at landfill. It was concluded that the best way forward concerning the waste at landfill was to introduce MBT as a pre-treatment step followed by AD of organic waste. This can be simulated by the WROSE model scenario 4.

A major municipal waste generator was identified, and the disposal rates at Clairwood Fruit and Vegetable market was assessed. A characterisation of the waste at the market showed that on average the market generates 2560 tonnes of waste per annum. A good portion of this waste is sent for landfilling. A stream analysis showed that the F&V waste stream was composed greatly of cabbages, oranges, lemons, carrots, melons, potatoes, and onions.

A characterisation of a sample of waste from the market showed a high moisture content of 88% with a VS content of 93%. The waste had a proteins and carbohydrates content of 16.7% and 51.2% respectively and thus making it a good feedstock for AD. However, due to the high acidity of the waste, it could not be digested solely. A co-digestion feed of CD and F&V waste was used.

The parameters investigated were pH, co-digestion ratio as well as pre-treatment ratio at a mesophilic temperature. A set of reactors were buffered at substrate pH, the second was unbuffered, whilst the third was buffered but the pH lowered to 6.5. The results obtained showed the optimisation of the digestion leaning towards high pH between 7.5-8.5. Buffered reactors generated the highest methane (101 L/kg VS) and were followed by unbuffered reactors, whilst the low pH reactors generated the least amount of methane. Furthermore, an 80:20 (CD: F&V) co-digestion ratio yielded a slightly higher methane compared to a 60:40 ratio. Pre-treatment was found to be very important with benefits of reducing particle size to <1mm resulting in more than double the methane generated compared to a particle size of 1-2mm.

The digestate from the process was analysed and the results showed that on average, the C/N ratio was lower than the optimally defined ratio of 20-30:1 which is important for soil improvement. It was at 11:1. The metal analysis showed the presence of N, P, K, Si, Nb, Br as important constituents returned to the soil. The digestate was furthermore, pyrolyzed and the product stream analysed. The pyrolysis increased the carbon content of the digestate by 12%. The biochar measured a carbon content of 39% compared to the 27% in the digestate. The biochar obtained had a higher heating value of 11.75 MJ/kg at a pyrolysis temperature of 500^oC compared to 4.30 MJ/kg in the digestate. The C/N ratio was increased to 48.5 but it did not fall within the optimum value described above. The bio-oil was obtained at a higher quantity of 78% and 63% during the pyrolysis of digestate at 350 ^oC and 500 ^oC respectively. It was found to possess a high amount of phenols, ketones and carboxylic acids.

The following recommendations were drafted from the research

- A carbon supplement is recommended to the substrates to increase the C/N ratio to the optimum range
- It is recommended that source separation of waste be implemented as this will make it possible for the diversion of organic waste
- The waste from large food-chain outlet must not be taken to landfill but sent to an organic waste only facility
- The public must be educated on the importance of handling of waste and separation of waste at source
- A possible upscale using the outcomes of this research is recommended and can make use of resources such as the WROSE model

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8. APPENDICES

8.1 Appendix 1- Additional Figures

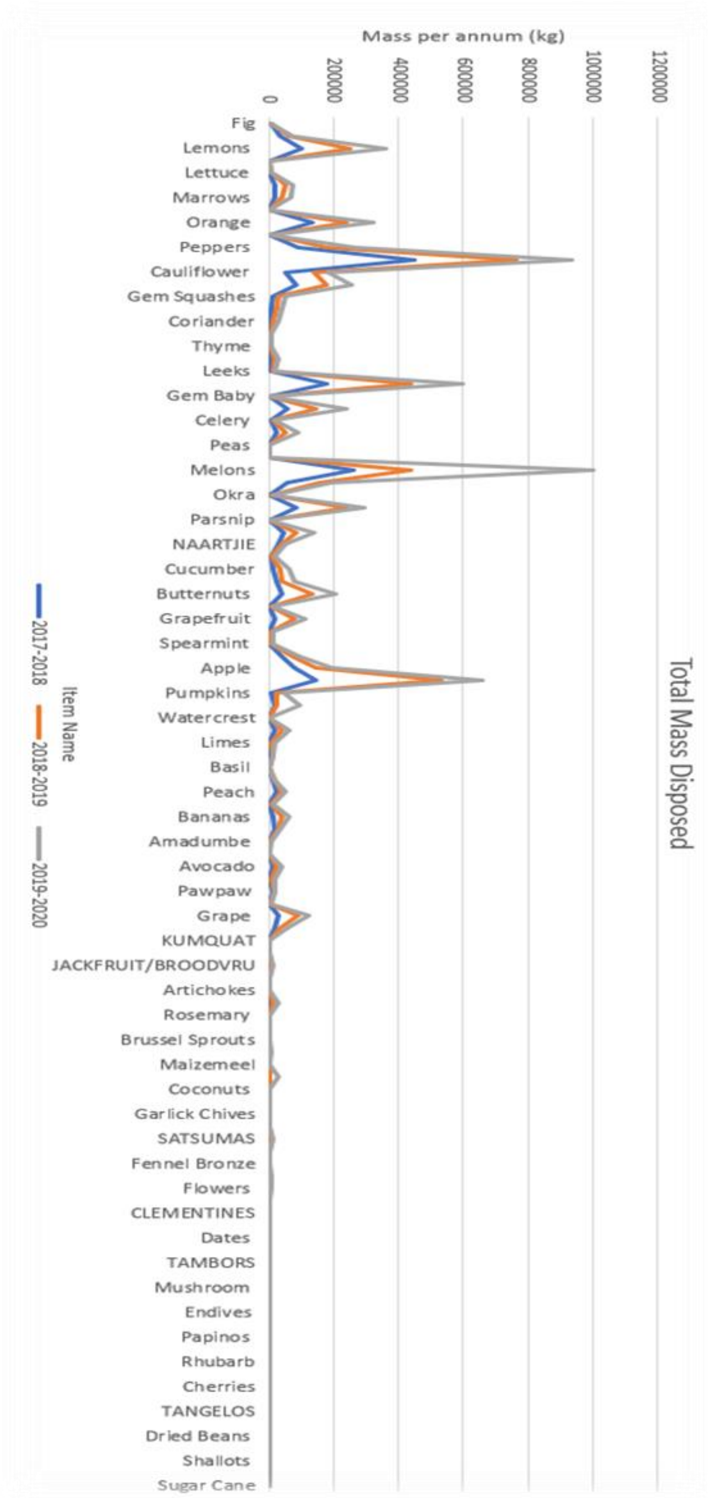

Figure A - 1: Fruit and Vegetable data at the Clairwood market

Appendix 8.2 Sample Calculations

1. Sample calculation for Proximate analysis (Dry Matter and Volatile Solids)

The data below is for Fruit and Vegetables waste

Mass of empty crucible (M_a) = 56.9265 g

Mass of crucible + wet sample (M_b) = 105.5428 g

Mass of crucible + sample after drying at 105 °C (M_c) = 63.5351 g

Mass of crucible + sample after Ashing (M_d) = 57.3233 g

DM (%) = $\frac{M_c - M_a}{M_b - M_a} \times 100\%$

DM (%) = $\frac{63.5351 - 56.9265}{105.5428 - 56.9265} \times 100\% =$ 13.6%

VS (%) = $\frac{M_c - M_d}{M_c - M_a} \times 100\%$

VS (%) = $\frac{63.5351 - 57.3233}{63.5351 - 56.9265} \times 100\% = 93.9\%$

2. Sample calculation for BMP experimental design

VS loading = 1.4 g/100mL solution

Working volume = 590 mL

VS required for substrate = $\frac{1.4\ g}{100\ mL} \times 580\ mL = 8.12\ g\ VS$

For a co-digestion mixture CD: F&V = 60:40

F&V = 40% of 8.12 g VS = 3.25 g VS

Actual dry mass added = $\frac{3.25}{0.939} = 3.46\ g$

The calculation for CD was done in the same way.

3. Conversion of methane volume reading to standard conditions

The following calculation is based on the 60:40 (<1mm) data

Measurement of methane volume (V_s) = 47.2 mL

Daily Temperature (T_m) = 296.15 K

Mean Atmospheric pressure (P_m) = 765.81 mmHg

Standard Temperature (T_{stp})= 273.15 K

Standard Pressure (P_{stp})= 760 mmHg

Methane volume at standard conditions = $V_{stp} = V_s \times \frac{T_{STP}}{T_m} \times \frac{P_m}{P_{STP}}$

$$= 47.2 \times \frac{273.15}{296.15} \times \frac{765.81}{760} = 43.87\ mL$$

4. Conversion of corrected methane volume to L/kg VS

Methane volume = 43.87 mL

Volatile Solids for substrates (VS) = 8.12 g

Methane yield (L/kg VS) = $\frac{43.85}{8.12}$ = 5.40 L/kg VS

5. Calculation of reactor volume

$$V_d = \frac{1000G}{YS}$$

G = 101 L/kg × 0.00812 kg VS = 0.0820 m^3 = 0.0328 m^3/day

Y = 9.33: see Table A-5

S = 23% × 1920000 = 441600 kg = 1.209 m^3/day

$$V_d = \frac{1000 \times 0.0328}{9.33 \times 1.209} = 2.908 m^3$$

8.3 Appendix 3 – Raw Data

8.3.1 Biomethane Potential Raw Data

Table A - 1: Biomethane potential raw data

V1- Adjusted pH, V2-Buffer, V3-No Buffer, V4-Blank (L/kg VS)

Day	V1	V2	V3	V4	V1	V2	V3	V4	V1	V2	V3	V4
2	0,75	5,40	1,54	0,12	0,59	1,30	2,25	4,69	0,41	1,93	0,67	0,99
3	1,67	11,33	3,75	0,97	0,76	1,88	2,61	5,75	1,42	4,28	1,52	1,61
5	4,27	23,17	11,45	4,70	1,50	3,08	3,38	7,21	3,14	8,54	3,92	4,06
6	6,82	36,42	12,12	8,25	2,76	7,33	4,67	9,08	4,02	12,73	7,69	5,16
7	11,98	51,13	16,85	12,42	3,45	13,27	5,64	9,70	4,58	16,64	9,08	6,24
8	21,95	66,45	23,47	13,94	4,41	17,03	5,91	10,89	5,38	21,52	10,68	8,48
9	32,10	73,37	30,36	14,58	5,63	19,64	6,67	11,83	6,79	25,75	14,33	11,18
12	39,11	83,55	40,46	16,31	7,67	22,90	7,52	13,93	7,63	27,09	16,36	12,86
13	45,13	90,57	48,54	17,15	8,83	25,01	8,04	15,02	9,28	30,51	20,06	15,75
14	48,84	92,94	58,59	18,42	10,30	29,27	8,81	17,57	10,50	33,67	23,57	17,79
15	49,01	93,85	65,43	24,08	10,26	31,20	8,84	19,08	12,17	36,13	24,31	19,30
20	51,75	94,91	67,00	31,66	10,97	36,16	9,76	21,27	15,94	41,24	31,27	23,41
21	54,16	98,18	69,53	38,96	11,90	40,38	10,35	22,68	17,26	44,34	35,59	25,13
22	57,35	99,17	70,12	41,74	12,45	43,63	10,76	25,08	19,27	50,53	41,27	28,14
23	58,25	100,07	70,77	43,22	12,69	47,24	10,92	29,90	23,65	54,59	46,67	35,37

8.3.2 Clairwood market Characterisation Raw Data

Table A - 2: Monthly data for Fruit and Vegetable Waste at the market

Month	2017-2018	2018-2019	2019-2020	Average
Jan	217649,3	235033,4	282950,3	245211
Feb	387279,5	341562,6	248255,05	325699,05
Mar	205655,9	449996,8	413988,15	356546,95
Apr	329529,4	241635,1	174506,05	248556,85
May	65840,5	259207	141771,55	155606,35
Jun	210663,1	129941,7	158129	166244,6
Jul	98023,1	73748,7	156048,8	109273,5333
Aug	123193,4	100413,9	108028,5	110545,2667
Sep	100853,28	152864,9	134630	129449,3933
Oct	231657,74	261772,8	123078,5	205503,0133
Nov	140119,5	118783,45	172754,35	143885,7667
Dec	290076,25	441756	358992,2	363608,15

Table A - 3: Yearly data for Fruit and Vegetable Waste at the market

Item name	2017-2018	2018-2019	2019-2020	Average annum
Fig	6795	0	0	2265
Sweet Potatoes	35408	25549	13611	24856
Lemons	102115	149238	106806	119386
Swedes	973	1561	2022	1519
Lettuce	2650	2355	1455	2153
Brinjals	17765	34076	22192	24678
Marrows	20787	18811	28951	22849
Hubbard	220	1014	1306	847

Squashes				
Orange	136142	103313	80980	106811
Patty Pans	3093	4203	2670	3322
Peppers	92219	88586	86662	89155
Cabbage	446770	318323	168274	311122
Cauliflower	49075	91366	43541	61327
Spinach	82524	97864	77398	85928
Gem Squashes	11083	16993	22812	16963
Danja	8163	15570	17330	13688
Coriander	5603	9354	13241	9399
Turnips	2770	3246	3602	3206
Thyme	1963	2308	2387	2219
Parsley	6526	12847	9125	9499
Leeks	6593	4928	5474	5665
Onion	178598	258946	159127	198890
Gem Baby	283	29	273	195
Calfornian Salad	56764	88895	91002	78887
Celery	7480	4970	7522	6657
Chillies	23208	26911	39040	29719
Peas	1320	837	911	1023
Rocket	481	1047	0	509
Melons	263447	172917	566065	334143
Broccoli	56862	97883	43068	65938
Okra	1243	5432	5407	4027
Carrots	83894	158121	50696	97570
Parsnip	18	71	0	30
Beans	48299	33915	56170	46128
NAARTJIE	32128	10215	11219	17854
BERRIES	7439	645	8509	5531
Cucumber	12870	21632	30756	21753
Pear	26561	14044	35897	25501

Butternuts	40882	91448	72267	68199
Greenmealies	3660	5717	3730	4369
Grapefruit	18950	59445	35985	38127
Radish	2548	3698	4148	3465
Spearmint	2265	5581	6233	4693
Beetroot	39610	34350	19948	31303
Apple	76989	69775	49271	65345
Potato	143434	385685	129550	219556
Pumpkins	9706	14628	23460	15931
Pineapple	17726	5630	71079	31478
Watercrest	396	698	488	527
Tomatoes	17317	23064	24380	21587
Limes	3832	3822	11132	6262
Sweetcorn	4172	6061	4541	4925
Basil	101	1553	355	670
Apricot	15343	2944	2764	7017
Peach	21781	20835	9790	17468
Garlic	1491	7425	4194	4370
Bananas	13900	25060	22406	20455
NECTARINE	21044	6326	5406	10925
Amadumbe	1379	1573	2838	1930
Nuts	275	512	795	527
Avocado	10531	14121	14414	13022
PERSIMMONS	4478	3107	9554	5713
Pawpaw	15559	1257	1238	6018
Curry leaves	796	1500	3026	1774
Grape	30101	61910	29634	40548
Plum	20216	15001	24642	19953
KUMQUAT	1094	48	0	381
Salad	155	630	570	452
Jackfruit/Broodvru	610	6068	5997	4225

Chives	58	101	0	53
Artichokes	140	30	0	57
Mango	5901	4458	17827	9395
Rosemary	161	413	260	278
Litchi	1080	802	284	722
Brussel Sprouts	80	0	10	30
Ginger	555	3532	2373	2153
Maizemeel	2040	803	0	948
Pomegranates	1383	2848	23022	9084
Coconuts	63	1253	702	673
Parsley Italian	84	20	0	35
Garlick Chives	50	0	0	17
Oregano	13	0	0	4
Satsumas	441	6734	6400	4525
HorseRadish	39	0	0	13
Fennel Bronze	16	18	50	28
Novas	2421	1208	3137	2255
Flowers	1790	1130	2082	1667
Guavas	247	1024	658	643
Clementines	1828	1047	1100	1325
Prunes	60	0	0	20
Dates	84	54	197	112
Quinces	1100	140	0	413
Tambors	438	770	1065	758
Granadillas	7	235	422	221
Mushroom	0	168	510	226
Honey	0	109	0	36
Endives	0	390	0	130
Fennel Green	0	28	0	9
Papinos	0	894	606	500
Chicken	0	158	0	53

Rhubarb	0	2	0	1
Sou-Sou	0	5	15	7
Cherries	0	376	0	125
Soup	0	490	0	163
Tangelos	0	0	1076	359
Minneola	0	0	440	147
Dried Beans	0	0	1220	407
Sprouts	0	0	2	1
Shallots	0	0	250	83
Kohl Rabi	0	0	80	27
Sugar Cane	0	0	20	7

8.3.3 Landfilling Raw Data

Table A - 4: Waste disposal data at the major landfills in eThekwini Municipality (Durban)

	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec	Total	Averages
01 - DSW	53950	47335	50965	47518	48017	44509	46208	47947	47649	52060	51557	52922	590638	49220
02 - GENERAL SOLID WASTE	11393	11874	12536	11926	13388	12380	11805	11760	11072	12820	13146	10803	144902	12075
03 - GARDEN REFUSE	4831	4902	4954	4625	4973	4553	4466	4483	4549	5329	4995	3945	56605	4717
04 - BUILDERS RUBBLE	6438	9173	10195	8836	10551	11144	9088	8587	8295	8671	8516	5314	104810	8734

05 - MIXED LOADS	1197	1257	1375	1229	1377	1305	1293	1339	1282	1474	1444	1240	15812	1318
06 - SAND & COVER MATERIAL	27271	32268	37844	34602	35925	34626	34639	37118	34660	40197	42318	21833	413302	34442
08 - TYRES	109	140	131	116	151	127	122	119	107	136	127	117	1503	125
09 - LIGHT TYPE REFUSE	47	39	50	45	48	58	50	51	50	56	47	49	589	49
10 - OTHER - Special Disposal	3312	3185	3370	2608	2506	2196	2316	2218	2536	2905	2786	2683	32622	2718
07 - PURCHASE COVER MATERIAL	3383	3357	3934	3819	5272	5021	2912	5222	2285	2481	4024	3512	45222	3769
25 - Imported cover material	2	6	7	6	8	9	5	4	7	6	3	0	62	5

8.4 Appendix 4 – Additional tables

Table A - 5: Yield factors for biogas production based on temperature and residence time (IRENA, 2016)

Feedstock retention time (in days)	Temperature (°C)					
	16-18	19-21	22-24	25-27	28-30	31-33
6-10	5.41	7.98	10.83	13.59	15.91	18.33
11-15	4.73	6.79	8.99	11.09	12.88	14.74
16-20	4.21	5.90	7.68	9.37	10.82	12.32
21-25	3.79	5.22	6.70	8.11	9.33	10.59
26-30	3.44	4.69	5.95	7.15	8.20	9.28
31-35	3.16	4.25	5.35	6.39	7.32	8.26
36-40	2.91	3.88	4.86	5.78	6.60	7.44
41-45	2.71	3.58	4.45	5.27	6.02	6.77
46-50	2.53	3.32	4.10	4.85	5.53	6.21
51-55	2.37	3.09	3.81	4.49	5.11	5.74
56-60	2.23	2.89	3.55	4.18	4.75	5.33
61-65	2.10	2.72	3.33	3.91	4.44	4.98
66-70	1.99	2.57	3.13	3.67	4.17	4.67
71-75	1.89	2.43	2.95	3.46	3.93	4.40
76-80	1.80	2.30	2.80	3.27	3.71	4.15
81-85	1.72	2.19	2.66	3.10	3.52	3.94
86-90	1.65	2.09	2.53	2.95	3.34	3.74
91-95	1.58	2.00	2.41	2.81	3.19	3.56
96-100	1.52	1.92	2.31	2.69	3.04	3.40

Table A - 6: Estimated diameter of a fixed dome plant (hemisphere design) by digester volume (IRENA, 2016)

Diameter (m)	Total plant volume (m^3)	Digester volume (m^3)	Gas storage volume (m^3)	Rated daily gas production (m^3/day)
2.0	2.09	1.68	0.42	0.70
2.2	2.79	2.23	0.56	0.93
2.4	3.62	2.90	0.72	1.21
2.6	4.60	3.68	0.92	1.53
2.8	5.75	4.60	1.15	1.92
3.0	7.07	5.65	1.41	2.36
3.2	8.58	6.86	1.72	2.86
3.4	10.29	8.23	2.06	3.43
3.6	12.21	9.77	2.44	4.07
3.8	14.37	11.49	2.87	4.79
4.0	16.76	13.40	3.35	5.59
4.2	19.40	15.52	3.88	6.47
4.4	22.30	17.84	4.46	7.43
4.6	25.48	20.39	5.10	8.49
4.8	28.95	23.16	5.79	9.65
5.0	32.72	26.18	6.54	10.91

Note: these figures assume that the gas storage volume is 20% of total volume and that it can hold 60% of daily production.