Intercepting Landfill Leachate for Recirculation:
An Initial Assessment of Experimental Design Parameters

By

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ABSTRACT

The primary goal of this research was to begin an understanding of the differences in using leachate removed at different depths in a landfill for leachate recirculation, on the biodegradation of waste.

The central question to be answered is:

> "Can any benefit be measured in using leachate for recirculation from shallower depths compared to the bottom of a deep landfill?"

10-year old waste from Kate Valley landfill in Canterbury, New Zealand was used in a laboratory experiment that simulated a landfill by using two large test cells with waste at different densities. It was expected that using waste of this age would ensure the waste was in a methanogenic phase of degradation, and methane composition and volumes would be sufficient to measure relevant properties.

By intercepting the leachate and using it for leachate recirculation and comparing this to leachate allowed to travel through the entire simulated waste pile, it was hoped that the various properties measured would demonstrate a difference in degradation efficacy.

The experiment failed to produce any gas volumes in any of the test cells. Post-mortem analysis to measure biomethanation potential showed there was little readily degradable organic material available for methanogenesis. The design parameters outlined in the research design section (section 3.3) had value and will help design future experiments. Measuring biomethanation potential via tube anaerobic respirometer is an effective means to determine likely successful gas production prior to committing to longer term landfill simulation experiments. Differentiating between slowly biodegradable organics from readily biodegradable organics during the experiment by measuring BOD, sbCOD and rbCOD may help to further clarify if readily biodegradable organics are available in the waste for methanogenesis.

This study also highlighted some interesting aspects of Kate Valley waste:

- There is significant evidence that much of the highly biodegradable organics expected in the 10-year old waste appear to have been degraded and can no longer support sufficient methanogenic activity for gas production.
- Phosphate may be a limiting nutrient in older waste at Kate Valley despite theoretical requirements indicating otherwise.
- Waste is considered to be a large particle sand, and as it degrades is more like a fine sand/silt. The particle sizing distribution shows that this assumption of waste is valid for the <20 mm distribution of this waste.
- A decrease in hydraulic conductivity with density was experienced. Density approaching 1600kg/m^3 experienced visible ponding although as waste is a highly heterogeneous material this is not a definitive density at which hydraulic issues can be said to occur within a landfill.

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1. INTRODUCTION

1.1. History

Rubbish disposal areas, variously described in historical records as middens, rubbish heaps, tips, dumps and landfills have always been part of our human society. Pohland, Cross et al. (1993) outline a history of waste from 3500 BCE, in Mohenjo Daro, part of the Indus Valley, Pakistan, when covered conduits were used to rot down waste. Rubbish is a regular source of archaeological evidence, especially of eating habits, and disposal areas were rarely referenced beyond sources of odour, until Colonel George E. Waring, given the task of cleaning up New York in 1894, "institutionalised and regulated the idea" [of recycling]. (Humes, 2012).

The concept of landfill has undergone significant changes over the last century due to direct factors such as the increase in quantities and varieties of wastes, and indirect factors such as a greater public awareness of the potentially detrimental effects of landfill on the local environment. This has forced changes in landfill practices. Early 19th-century waste management involved the dumping of collected municipal refuse in an area far enough away from town that sight and smell were not an issue. Waste was also raw material for land reclamation in areas such as swamps and wetlands. No restrictions on waste type were in place, and all manner of waste was disposed of together. Canterbury was not atypical in having shallow and small dumps with little waste compaction.

During the 1920s and 1930s, concerns over the health threats posed by dumping practices led to the evolution of the sanitary landfill. Sanitary landfilling began with placing a cover of clean soil over the garbage daily to decrease surface decay, minimise odour, vermin and fire risk and for aesthetic reasons.

Calvert (1932) in the Journal of American Water Works Association outlines contamination of a groundwater well 150 m away from "garbage liquor" in 1932. McBean, Rovers et al. (1995) pinpoints general recognition of potential of groundwater contamination by the industry to 1954. In the 1960s, episodes of contamination of local environments by landfills led to further developments in landfill management especially by the USEPA (Weaver, 1961). The concept of the engineered landfill was developed whereby waste disposal sites were lined with an impermeable layer to prevent the release of liquid emissions into the surrounding soil and groundwater. The Friends of the Earth's Citizens Guide to Municipal Landfills state most landfills built prior to 1993 do not have liners (foe.org, 2000). Since these early improvements the use of vitrification, bioreactors, anaerobic digesters, and other management practices such as waste segregation, landfill monitoring, and landfill mining have been used to further minimize the impact of landfills on the surrounding environment.

1.2. Leachate

The liquid resulting from waste is called leachate. It can be produced directly from liquid wastes or fluid-bearing wastes that are part of the waste stream or by dissolution or decomposition of inorganic and organic constituents in the waste and the reactions aided by water (and other liquids) passing through the waste.

Boyle and Ham (1974) state that prior to 1965 landfill leachate was unrecognised in the public arena, (although it had been recognised internally in the waste industry since 1932) and few cases of leachate causing concern were reported. It was assumed that natural soil under a landfill would attenuate or purify the leachate. There is some merit to this way of thinking as soils have natural attenuation properties including:

- Filtration
- Physical sorption
- Precipitation and dissolution
- Complex chemical reactions
- Catalytic activity
- Biological degradation
- Redox potential
- Ion exchange capacity.

Leachate leakage is now accepted as one of the highest risk factors in a landfill. To mitigate this risk design philosophies consider geology, the siting of the landfill, the use of drainage layers, the placement of liners and consideration of waste geotechnical properties. Zeiss (1992) showed that leachate flow is characterised by gravitational layer flow on solid surfaces and it is this flow of leachate that flushes the waste, eventually collecting at the lowest point, the base of the landfill.

Guidelines such as the International Solid Waste Association Landfill Operational Guidelines (ISWA, 2019) advocate minimising leachate accumulation to assist in maintaining leachate head below certain levels even after closure. This minimises the driving force of leakage through the base and maintains mechanical strength of sidewalls. Removal of leachate from landfills is vital and often regulated e.g. by USEPA regulations or as part of licencing of landfills in New Zealand.

1.3.   Landfill Stabilisation

Waste disposal volumes to landfill is constrained by landfill capacity and by after-care requirements. Optimal lifecycle management for landfill owners is to make the best use of the capacity available and ensure post-closure risk is mitigated at optimum cost.

Moisture within the waste, along with water from precipitation infiltrating the waste, flushes and stimulates oxidation and biological activity, effectively degrading the waste. This phenomenon has been exploited by humans for millennia as evidenced by walled bioreactors in Kouloura, Crete c.a. 1900 BCE (Pohland, Cross et al., 1993). Degrading waste, along with dissolution, the physical action of the liquid flow-through, and microbial activity, settle and stabilise a landfill over a long period. This settlement has been measured in the laboratory. For example, Kadambala, Powell et al. (2016) measured settlement in situ over five years at an accuracy of ±10 mm. Fei and Zekkos (2013) presented a literature review of settlement data from 98 tests over 29 mesoscale-simulators and landfills.

Reinhart and Townsend (1998) in their Landfill Bioreactor Design and Operation text assert that encouraging fast waste stabilisation is key to landfill bioreactor success. McBean, Rovers et al. (1995) summarised the advantages in hastening waste stabilisation to include:

- Earlier degradation of waste
- Intensification of methane gas production, improving gas utilisation efficiency
- Faster leaching of contaminants
- More predictable liquid discharge volumes and quality
- Increased available volume resulting from airspace created during settlement
- Achieving the intended final use of the landfill occurs earlier.

Laboratory, pilot-scale studies and full scale experiences like those reviewed by Kulkarni and Reddy (2012) and Reinhart and Townsend (1998) have shown that control of moisture enhances stabilisation of waste. Stanforth, Ham et al. (1979) and other researchers following them point to biological processes as the most significant degradation process. Christensen and Kjeldsen (1989) described the processes whereby enhanced landfill degradation occurs by promoting anaerobic biological activity; key to this is supplying water and nutrients.

Water and nutrients are known to be already present in leachate. The recirculation of leachate back into the landfill has been accepted practice for several decades now (Pohland, 1975) and is known to speed up the stabilisation process. Leachate recirculation is not common as part of normal operation of landfills; this is usually due to cost and the operational effort required. The benefits of recirculating leachate in a broad sense is articulated in Appendix A. Advantages and disadvantages of leachate recirculation as summarised from landfill literature is summarised in Figure 1.

Increased methane production
Improved flushing and rinsing of waste
Improved chemical, nutrient and water transport mechanism
Enhanced biological activity leading to faster waste degradation
Buffered leachate volume
Delayed leachate disposal or treatment
Treatment for leachate BOD
Reduction of leachate volumes through evaporation and sorption

Increased solids in leachate collection equipment
Increased bioclogging
Odours
Capital and maintenance costs of equipment
Necessity to handle higher hydraulic loads
Increased consequences to leakage
Increased risk of anaerobic disruption due to pH, ammonia/ammonium, toxin levels
Risk of higher levels of leachate locally and generally

Figure 1 Advantages and disadvantages of leachate recirculation
(Sources: Townsend (2015), Chung, Kim et al. (2015), Sponza and Ağdağ (2004), McBean, Rovers et al. (1995), Qasim and Chiang (1994), Leuschner (1989) and Pohland (1975)

1.4.   Intercepting Leachate Flow

In the context of this thesis, the well-established science behind the recirculation of leachate and measurement of stabilisation are utilised. The present state of research on landfills contains an intrinsic assumption that leachate is collected at the bottom. Research pilot tests and laboratory apparatus are also designed to collect at the bottom with intercept collections only occurring incidentally for analysis purposes. The exception to this bottom-collection is wells inserted into failing landfills either to investigate or correct leachate flow and/or hydraulic problems. Vertical addition wells have been used to inject leachate into the landfill (Kadambala, Powell et al., 2016) and wells and/or trenches have been dug into landfills post-closure to either investigate or recover rectify leachate flow issues.

Conceptually intercepting leachate above the base of the waste pile for recirculation has the potential to reduce some of the disadvantages listed in Figure 1. Beginning an investigation to explore the impacts of recirculating intercepted leachate rather than bottom collected leachate is proposed to advance understanding of recirculating leachate mechanisms.

1.5.   Scope

The scope of this work is limited to consideration of in-situ leachate recirculation that could be applied at the Kate Valley Landfill (Canterbury NZ). In this regard, factors considered in other research such as stabilisation in arid climates, geotechnical strength, and freezing temperatures are not considered. Kate Valley Landfill has been in operation since June 2005, has an integrated leachate collection system and little on-site treatment facilities. Thus, aspects such as treatment of leachate, adding nutrients, buffers or other chemicals or microbes are similarly considered out-of-scope in this study.

1.6.   Thesis Structure

Chapter 2, the **Literature Review**, builds on this introduction to outline landfill and waste parameters and existing relevant knowledge about leachate to enable existing mathematical models and conceptual representations of waste stabilisation to be applied to the experiment.

Chapter 3 (**Research Design**) outlines the experiment in terms of research objectives to be achieved, basis for design, equipment and measurements. Chapter 4 outlines specific **methods** to obtain the results. This is followed by the results of the experiments in intercept leachate and biomethanation potential (supporting those outcomes) in Chapter 5 (**Results**).

Chapter 6 presents the **discussion** of the results in terms of the original hypothesis and objectives and outlines other considerations and further research that may assist the enquiry of intercept leachate. This is followed by an overall **conclusion** and detailed **appendices**.

2. LITERATURE REVIEW

2.1. Composition of Waste to Landfill

2.1.1. Categorisation of Composition

Communities tend to consider solid waste as two streams: Municipal solid waste (MSW) and construction and demolition (C&D) waste. Both streams can contain similar materials, but MSW is characterised by a high number of discrete acts of disposal making it more homogenous. Table 1 outlines some key differences between these two waste streams.

Table 1 Characteristics of solid waste streams

Characteristics of waste	MSW	C&D
Frequency	Frequent	Infrequent
Size	Smaller	Larger
Quantity	Large numbers	Small numbers
Life	Short	Long
Homogeneity	More homogenous	Less homogenous
Organic content	Higher	Lower

There is also a third type of waste known as industrial waste. Industrial waste tends to be of a composition specific to the industry and dealt with on a case by case basis. The type of waste (MSW, C&D, industrial) governs its pre-treatment, disposal method and management practices. Municipal waste is compacted and deposited directly into a MSW landfill, C&D waste can be part of MSW landfill or landfilled directly as "clean-fill". Industrial waste may require stabilisation followed by placement and is not considered further herein.

Cultural tastes, societal behaviour, waste collection and recycling processes, locality and temporal factors all affect MSW waste volume and waste composition. Different countries and even different landfills therefore have different waste characteristics. Waste is a heterogeneous material, its properties change spatially and over time. Landfilled waste composition has substantially altered over the last few decades. Letcher and Vallero (2019) outline temporal changes over decades as well as seasonal changes, describing differences and similarities between developed and developing countries and the differences between affluent and poor areas. Letcher and Vallero (2019) and Campbell (1989) highlight the disposal of much higher proportions of potentially degradable, organically based materials, simultaneously with an increase in plastics and a subsequent decrease in proportions of metals and textiles. Parallel to this is the influence of recycling initiatives removing organics, metal and selective plastics from the MSW stream.

Waste composition is the most influential parameter controlling the characteristics of a landfill. It affects biochemistry, leachate composition, gas production and stabilisation. For example, gas production volumes differ by a magnitude between vegetables and wood products.

When looking at waste composition, categorisation of waste into generic classes also differ. For example, "green" waste can be alternately labelled as food, organic, putrescible, vegetative, food and garden, and may contain either food or garden waste or both. Terms are not clearly defined and there is no international standard methodology for characterizing waste (Dahlén and Lagerkvist (2008), Edjabou, Jensen et al. (2015)). This adds a layer of uncertainty and complexity to literature comparisons however, considering ranges rather than absolute numbers assists with understanding the influence of various categories.

2.1.2. Composition Basis

The range of solid waste compositions based on research data consulted throughout this thesis is indicated in Figure 2. The references and broader context appears in Appendix D2 – Waste Characteristics.
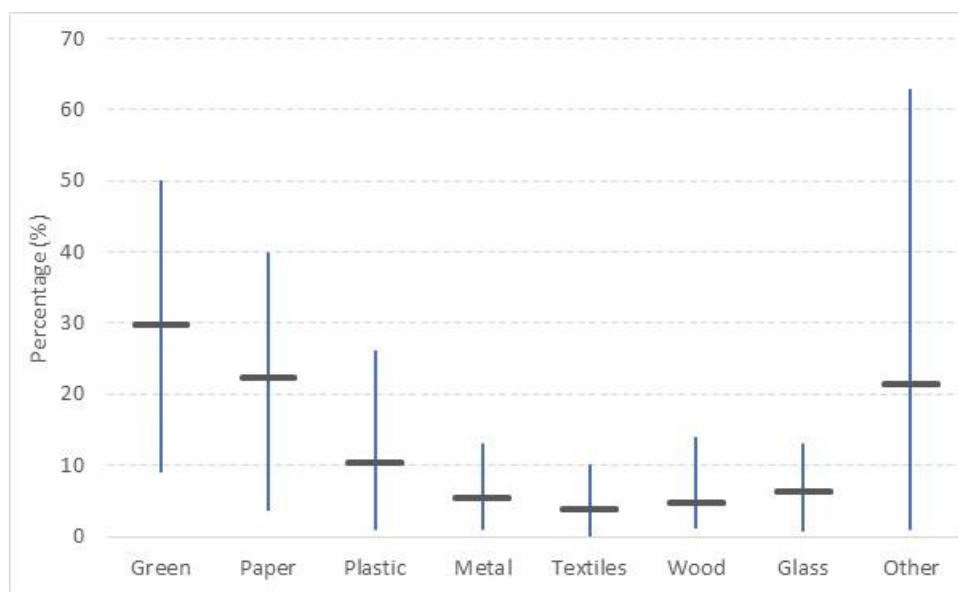
Figure 2 Waste composition means and range based on previously published research

2.1.3. New Zealand Context

Perrot and Subiantoro (2018) compiled an overview of waste composition in New Zealand landfills which falls within the ranges above (Figure 3).
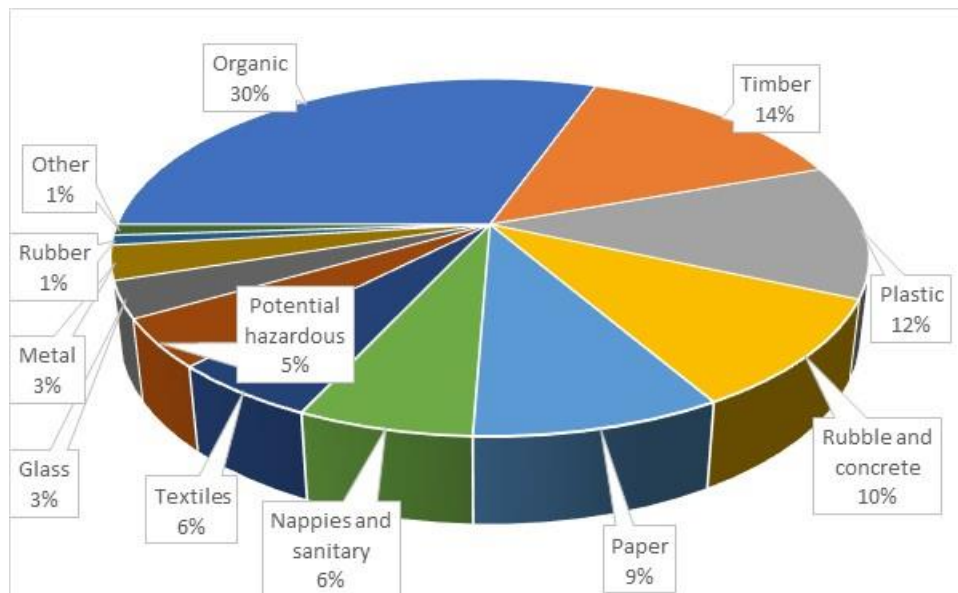
Figure 3 Waste composition of landfills in New Zealand between 2011 and 2017 (adapted from regional council reports as compiled by Perrot and Subiantoro (2018)

2.1.4. Chemical Composition

A valuable paradigm described by Milke (1992) sets out the composition of waste from a chemical perspective as if it is a meal to be enjoyed by the microbes. This "bacteria-based description" allows comparisons of waste to be made outside the normal consumer-use basis of paper, wood, metal, glass, food scraps and plastic.
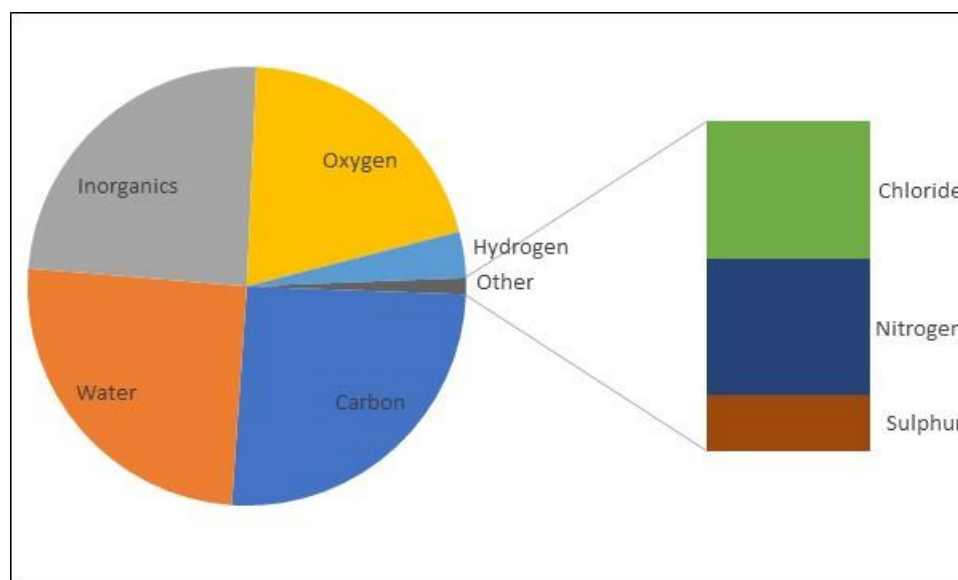
Figure 4 Chemical composition of MSW landfill waste
Source: Pohland, Cross et al. (1993)

This representation better illustrates that the levels of carbon, water, inorganics and oxygen are similar and other elements occur as a fraction of these levels. This is an important consideration when assessing bioactivity since carbon, or organic material may be present in the waste stream but may not be accessible or readily consumed. For example, simple carbohydrates like sugars present in many food wastes are more easily broken down by bacteria than celluloses present in wood or paper.

2.2. Variability in Data

Properties can be difficult to measure especially in landfills, and published data is by its nature limited and often unclear or incomplete in describing methods, uncertainties or circumstances around measurement.

The nature of the interactions within landfills and within the microbial environment can usually only be inferred rather than directly demonstrated. Durmusoglu, Sanchez et al. (2006) note the difficulty of laboratory testing with sampling under anaerobic conditions and the need for large-size samples. Sampling and sorting procedures influence waste composition measurements and, while the literature outlines good sampling and sorting processes, many researchers do not detail methods used. Reviews of waste literature such as performed by Gettinby, Sarsby et al. (1996) and others outline some of this variability. It is worth noting that since there are no generally accepted sampling and testing procedures for waste materials, the principles of general soil mechanics are usually applied, often leading to variation in methodology and results.

It is clear that landfills are highly variable in many aspects. The comparison of results from literature have an extra dimension of complexity in that not only is waste spatially and temporally variable but research also alters waste properties through deliberately altering sizes, density or moisture content to suit experimental aims: Some researchers such as Reddy, Hettiarachchi et al. (2011) prepare laboratory and pilot studies with synthetic waste, some use reconstituted waste (Francois, Feuillade et al., 2007) or shredded waste (Barlaz, Milke et al., 1987), some work with raw waste such as Sel, Çakmakcı et al. (2016) and Dixon and Jones (2005) and other researchers work within the confines of the landfill itself (Chung, Kim et al. (2015), Ehrig (1983)).

2.3. Landfill Stabilisation

Landfills stabilise through degradation of the waste via physical, chemical and biological processes. These processes occur in multiphase mediums (gas, liquid, solid, microbial) each interacting with the other phases and containing significant spatial and temporal variations. As explained in section 1.3, Stanforth, Ham et al. (1979) and other researchers following them, point to biological processes as the most significant degradation process and they control to a certain extent the chemical and physical processes.

Physical processes for degradation include:

- Rinsing
- Physical movement
- Structural change of waste (due to water contact or compression)

Chemical changes arise from:

- Oxidation
- Reduction
- Change in pH
- Dissolution
- Complexation

- Other chemical reactions

There are many representations of waste degradation depending on which aspect of the degradation processes the researcher is particularly interested in. A simple picture with which to start considers the degradation of waste in a landfill as occurring in a series of two phases: saturation followed by biological breakdown.

Saturation occurs from precipitation (rainfall, dew, snow) both when the waste is open and percolation through the soil day-cover and/or cap. Moisture is also present in or as part of the waste.

Biological activity is known to consist of a short aerobic period followed by prolonged anaerobic degradation of predominantly the organic parts of the waste. The actual microbial processes for biological activity is complex, and they are represented in various ways, depending on research aims. Frequently the anaerobic phase is split into an acidogenic and methanogenic phase. Figure 5 illustrates as a cascade these simplified descriptions. These concepts of degradation have developed over the years to more complex explanations but this representation provides an adequate picture for understanding and discussion.
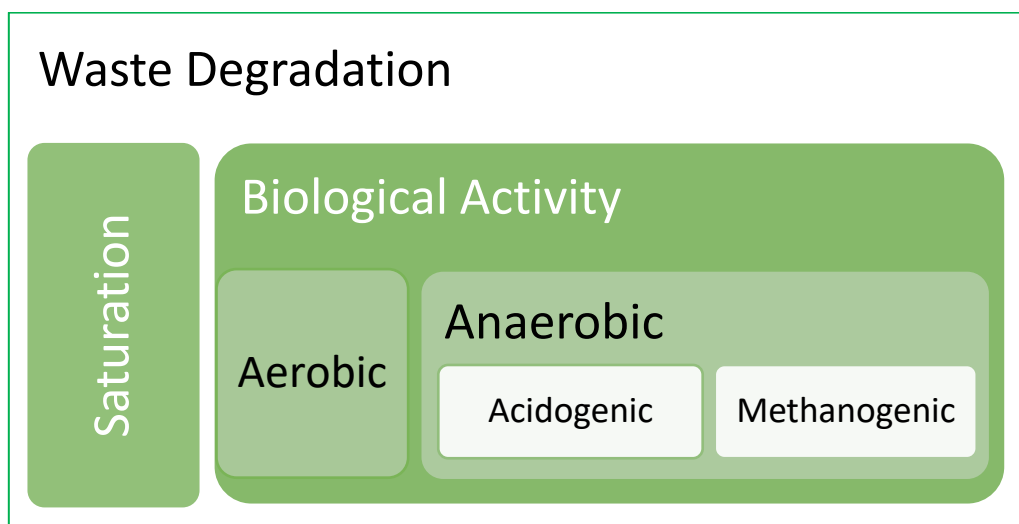
Figure 5 Simplified overview of the degradation of waste

Aerobic degradation of organic matter occurs predominantly in fresh waste and in old non-compacted landfill. Aerobic degradation requires oxygen, therefore is limited to fresh waste just below the surface of an active landfill, air transfer deeper within a landfill is limited, making aerobic degradation short-lived in these areas. Aerobic decomposition degrades organic material to carbon dioxide, water, and partially degraded organics producing heat. When the oxygen supply is exhausted the anaerobic microorganisms take over. In modern, large, and deep landfills, waste undergoes compaction as it is deposited. This reduces aerobic potential further and results in decomposition primarily by anaerobic processes. Although the overall process is sequential, it is not relevant to think of them occurring one at a time. In the current view of how degradation works, the biological activities as well as the physical and chemical processes occur simultaneously, but they occur in local micro environments with each going through the processes at different rates and with different yields.

2.4.   Biochemical Explanation

2.4.1. Phases of Anaerobic Degradation

The production of methane in landfills depends on a mixture of microorganisms that degrade the complex organics in sequential but not wholly independent phases. DeWalle (1978), Barlaz, Milke et al. (1987), Bareither, Wolfe et al. (2013), Madigan, Martinko et al. (2015), Majdinasab, Zhang et al. (2017) and others refer to these phases as hydrolysis, fermentation, acetogenesis, and methanogenesis. Figure 6 shows this sequence (based on data from laboratory scale lysimeters) with the main products of each step.

Hydrolysis	Fermentation	Acetogenesis	Methanogenesis
•Soluble monomers	•Short chain VFAs, CO_2, hydrogen	•Acetate, hydrogen, CO_2	•Methane, CO_2
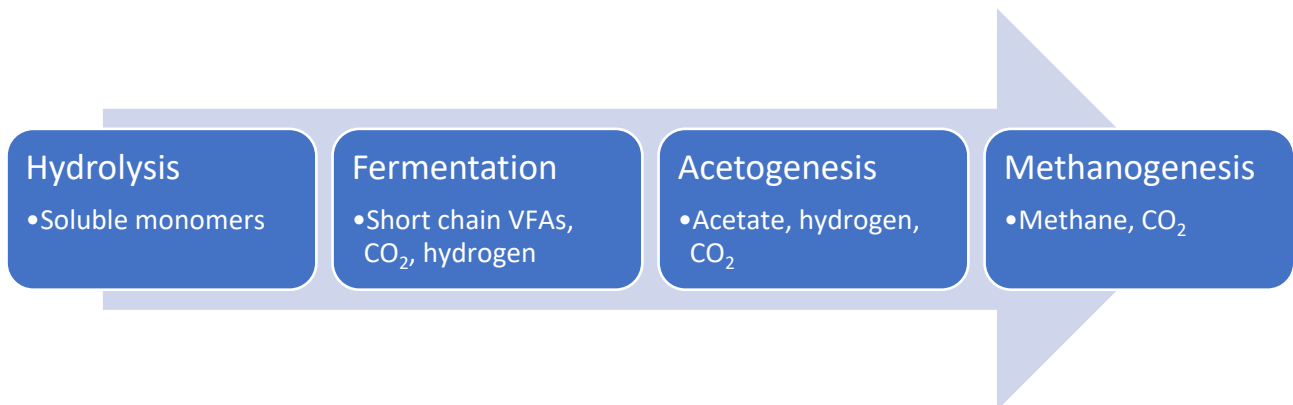
Figure 6 Key steps of the anaerobic degradation process
Adapted from: Waste360.com (2016).

There are many ways to represent the biochemistry in landfill degradation and various representations have been developed to help understand aspects of chemistry, biological interactions, geotechnical properties and mechanical behaviour of landfills.

More detailed representations than those above have been used to provide further understanding of microbial and chemical degradation. Ehrig (1983) uses a theoretical sequence of five phases to define the degradation of waste in terms of gas production and leachate chemical parameters:

Phase I: Short aerobic phase that begins the degradation. Starts in fresh waste. Easily degraded organic matter is aerobically decomposed. Lasts days.

Phase II: First intermediate/transitional anaerobic phase: Fermentative and acetogenic bacteria. Quick process.

Phase III: Second intermediate/transitional anaerobic phase: Slow growth of methanogenic bacteria.

Phase IV: Methane phase: Characterised by stable methane production.

Phase V: Maturation: Low levels of more stable organics remain in the waste and microbial activity is very low.

Reinhart and Townsend (1998) describe the final maturation stage as stabilisation, discernible by the lower leachate volumes, lower leachate analyte concentrations, and the ramp down of gas volumes.

The phases can be understood in terms of the relative or changing concentrations of the various gases and compounds present in leachate. Other parameters such as pH, alkalinity, redox potential, oxygen demand, and metal precipitate levels can also support an interpretation of the degradation progress. Analysis of parameters such as these allows the progress of degradation to be understood and followed during the lifecycle of a landfill or during the simulation of degradation in the laboratory.

2.4.2. Temperature

Anaerobic activity is affected by temperature. Most methanogenic bacteria present in landfill are mesophilic and reproduce at a maximum rate around 40 ºC (Senior, 1995). While heat generation from anaerobic degradation is often ignored, it does yield about 7% of the heat generation of aerobic degradation; Campbell (1989) quotes 632 kJ compared to 9300 kJ based on glucose decomposition, although El-Fadel, Findikakis et al. (1997) maintain this heat may be underestimated due to the complexity of the large set of parallel and sequential reactions. Because of the size and insulating capacity of waste in a landfill situation, heat loss may be low and the heat generation created by anaerobic degradation has the potential to raise the temperature inside a landfill.

2.5.   Leachate Processes

2.5.1. Leachate Cycle

Leachate in landfills can be thought of as four distinct and cyclic processes:

1. Leachate generation
2. Leachate movement
3. Leachate collection and treatment
4. Leachate recirculation.

The four processes in this leachate progression can also be considered via the physical, chemical, and biological processes underway. Complexity arises in that the physical, chemical and biological processes interact with each other and contribute in part to leachate generation, movement, and recirculation. Figure 7 summarises areas for consideration in this cycle.
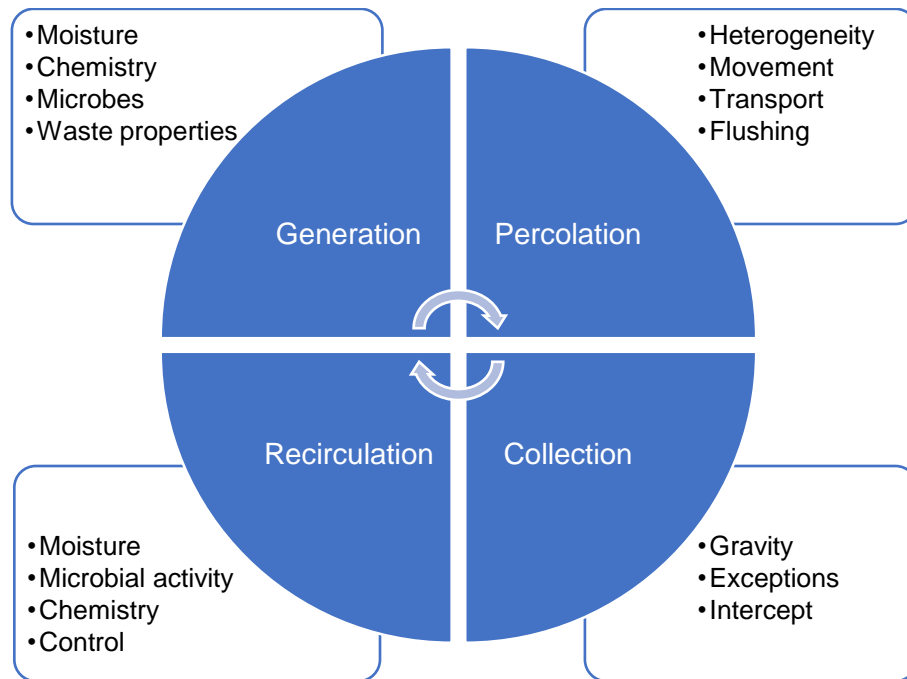
Figure 7 Leachate lifecycle processes

2.5.2. *Leachate Generation*

Leachate generation occurs as soon as there is excess water beyond reaction and absorption mechanisms. It occurs at a certain water content which can generally be assumed to be less than field capacity due to short circuiting and channelling effects through the waste layers (Canziani and Cossu, 1989). Texts such as Qasim and Chiang (1994) and Reinhart and Townsend (1998) and review papers like Leckie, Pacey et al. (1979) outline the factors that influence leachate generation. Typically, these have been determined by analysing water balance and chemistry of landfill products (leachate, gas, degraded waste) and include moisture sources and movement, chemical conditions, biological requirements and waste properties.

Qasim and Chiang (1994) note from previous researcher's studies that polluting chemicals characteristically occur at higher concentrations in deep fills, because the leachate has longer contact time and travels greater distances.

2.5.3. *Percolation of Leachate*

Leachate flows are delayed until field capacity of the waste is reached, although this may occur in localised regions within a landfill at different times. When the moisture content of the waste exceeds capacity, free-flowing liquid moves under gravity (Zeiss, 1992), flushes and rinses waste and acts as a transport mechanism for nutrients, bacteria, alkalinity and chemical inhibitors of microbes to other areas within the landfill (McBean, Rovers et al., 1995). The movement of leachate also allows for movement of small particles creating settlement and landfill stabilisation.

Ko, Yang et al. (2016) and Bolyard and Reinhart (2016) also support flushing as one of the functions served by leachate movement. Water alone was shown to serve the same purpose in a

similar manner. The addition of any moisture improves key biological reactions, however, leachate has the advantage of supplying microbes and nutrients. This leads to increased anaerobic decomposition rates in a complex interrelated fashion. Additionally, avoiding the addition of water decreases the volume of leachate that must ultimately need to be treated.

According to Zeiss (1992), the heterogeneity of landfills has a strong influence on hydrology, and as such, flushing is limited, as preferential pathways often exist. The flow of water through Municipal Solid Waste (MSW) landfills can therefore be considered highly non-uniform and dominated by preferential pathways.

2.5.4. Collecting Leachate

As mentioned earlier, Zeiss (1992) characterises flow in a landfill by gravitational layer flow on solid surfaces. This clarifies why leachate is almost exclusively collected from the bottom of landfills via a drainage layer. Collection systems are now common and made of multi-material composites of plastics, geosynthetics, sand, gravel and a network of pipes. Environmental engineering texts as well as specialist landfill books describe these systems, and some are referenced in this thesis, such as Townsend, Powell et al. (2015), Mihelcic, Zimmerman et al. (2014), Qian, Koerner et al. (2002), Reinhart and Townsend (1998), McBean, Rovers et al. (1995), and Daniel (1993). New Zealand has guidelines known as the Land Disposal Guidelines or Landfill Guidelines which can be sourced through WasteMINZ website.

Exceptions to bottom collection systems only appear to include investigative cores (Belevi and Baccini, 1989) or operational corrective wells inserted into landfills with failures in gas or leachate collection or landfills designed without collection, as described by McBean, Rovers et al. (1995) and Qasim and Chiang (1994). While these wells have been useful for further understanding landfill hydraulics, no information has been found to date to suggest they have been routinely utilised for determining leachate characteristics within the landfill volume.

There are hydraulic issues with collecting leachate at the bottom of a landfill. As leachate passes through the lower portion of a landfill it flows through more compressed and more degraded waste with higher density and lower hydraulic conductivity. This increases the likelihood of local pooling and leachate seepage, potentially creating local leachate accumulation. Additionally, bioclogging of the leachate collection system can also occur leading to leachate accumulation, increasing the risk of leakage and sidewall failure.

2.5.5. Leachate Recirculation

Numerous studies have demonstrated that decomposition of waste in landfill occurs more rapidly when the moisture content is increased (Pohland, 1975, Leckie, Pacey et al. 1979, Pohland 1980, Leuschner 1989, Christensen, Cossu et al. 1989, McBean, Rovers et al. 1995, Reinhart and Townsend 1998, Hossain, Gabr et al. 2003, Sponza and Ağdağ 2004, Laner, Crest et al. 2012, Fei and Zekkos 2013, Tolaymat, Kim et al. 2013, Bolyard and Reinhart 2016, Degueurce, Trémier et al. 2016, Ko, Yang et al. 2016, ISWA 2019, and Karimi and Bareither 2021).

The moisture available in the waste is usually not sufficient on its own to meet microbial requirements. Townsend, Powell et al. (2015) note that the volume of moisture available at a

landfill is insufficient to reach field capacity in a reasonable time frame, so design and operational modifications are needed if the intention is to add liquids to the landfill waste. Leachate is the most common liquid supply, but other moisture sources can also be used. Leachate recirculation, therefore, can be supplemented with the addition of other liquid sources such as water (surface water or groundwater), sludge from wastewater treatment plants, or other available liquid wastes. This practice is generically referred to as moisture or liquids addition.

Huang, Wang et al. (2016) showed that recirculation of leachate improved the microbial activities and the solubilization of organics in the landfill cell. They did this by measuring the compaction ratio (difference of the height of stabilized and original wastes in the cell) and organic/inorganic ratio as two indices for waste stabilisation.

Reinhart and Townsend (1998) showed, using studies of COD half-lives, that COD levels are reduced by five to ten times using recirculated leachate. This is based on their own and previous research by others as far back as Pohland (1975), from both laboratory studies and landfill data.

It is known that leachate, if continually recirculated without treatment or chemical modification, can interfere with and even stop bioactivity. The causes of this vary with the type of biological activity present, and have been shown to include altered oxygen demand, lower pH, and increase in toxic or inhibitory chemicals. Bacterial activity can be exposed to a variety of abiotic factors in the highly heterogeneous landfill system. Huang, Hui et al. (2004) believed that poorly-controlled leachate recirculation can cause inhibitory effects on the anaerobic process. McBean, Rovers et al. (1995) reported that for methane forming bacteria the variability in oxygen, hydrogen, pH/alkalinity, sulphate, nutrients, inhibitors, temperature and water content affects the ecosystem and consequently the rate of activity.This occurs over both the short- and long-term. Thus, control of leachate recirculation in terms of quality is necessary to ensure an optimised environment for bioactivity.

Degueurce, Trémier et al. (2016) and Karimi and Bareither (2021) have shown experimentally that for anaerobic digestion, recirculated volume of leachate and rate of application need to be matched to increase methane generation and decrease lag-time for microbial growth. There is no reason to expect that a similar process is not valid for landfills. Feng, Bai et al. (2018) report a field verified recirculation intensity of between 15 and 150 litres/ton of waste per annum. Leachate recirculation flow parameters such as volume and periodicity are therefore considered important to control.

There are also hydraulic considerations with continual leachate recirculation. Volume of liquid in the landfill will keep increasing, which increases hydraulic head and associated risks of leakage, sidewall failure etc. Therefore there may be a requirement to bleed off some leachate at certain times and treat it.

2.5.6. Leachate Treatment

Experience in leachate recirculation has shown that long-term landfills typically produce excess leachate (exceptions occur in arid regions or during dry spells). While this excess can be retained in the landfill to some extent, risk minimisation strategies demand keeping leachate head in the drainage layer at the base of the landfill to a minimum, requiring the leachate to be removed. It is therefore normal operational procedure to bleed off collected leachate and send it for treatment

and disposal as part of leachate control. Treatment (most often via wastewater treatment plant) is not part of this study, however it is worth mentioning that recirculation is a partial treatment process for leachate, as the landfill can be considered a type of anaerobic digester that can decrease organic carbon content and has the capacity to sorb toxins.

2.6.   Intercept Leachate

Rather than waiting for the leachate to travel through the entire depth of the landfill, intercepting it higher up in the waste pile results in the leachate being exposed to the various chemical and biological activities for less time. This has the potential advantage to reduce the quantity of solids entrained and acid, ammonium and odours dissolved in the leachate. It also has the disadvantage of reducing the rate of attenuation which can be a beneficial property for reducing hazardous or toxic compounds. Volume collected in the bottom of the landfill will also be less, lowering the potential for bioclogging and decreasing head and hence leakage risk. Opportunities for reduction of leachate volumes through evaporation may be enhanced through faster leachate recycle rates.

However, the study of intercept leachate appears to have no basis in literature and the opportunity to grow the knowledge in this area has value.

2.7.   Mathematical Models

The prediction of leachate generation has evolved from early empirical equations with limiting assumptions about moisture flow through to using artificial intelligence algorithms as outlined recently by Abunama, Othman et al. (2019).

Similarly, leachate quality prediction has progressed from early rate equations such as used by Lu, Eichenberger et al. (1984) to summarise the relationship between landfill age and some leachate compositions over a limited time range. For example, remaining BOD_5 levels were expressed as a first order rate equation limited to a period of 3-30 years as $C_b = C_{b_0} * 10^{-kt}$ where C_b is the remaining concentration at time t, and C_{b0} is the maximum concentration (assumed at t=0).

Qasim and Chiang (1994) summarise a plethora of research into leachate characterisation mathematical models to predict leachate quality and conclude that first order rate equations conveniently expressed the behaviour of upper limits of data for landfills aged 3-30 years. These first-order forms assume that there is an exponential relationship between the leachate constituent concentration in the waste and the decay rate. They are typically based on the form a * 10^{-kt} or a * e^{-kt}, where a and k vary by chemical. For example, Lu, Eichenberger et al. (1984) estimate COD levels for landfills 2-10 years old to be represented by the equation:

$$89{,}500 * 10^{-(0.0454)t} \ mg/L \qquad\qquad (1)$$

and organic nitrogen concentrations can be represented by:

$$130 * e^{-(0.185)t} \ mg/L \qquad\qquad (2)$$

Lu, Eichenberger et al. (1984) concluded that these rate equations were useful for organic type compounds but did not adequately explain heavy metal levels.

Sivakumar (2013) used hydraulic properties of solid waste as inputs for an analytical model of leachate flow. Using properties such as moisture content, field capacity, and hydraulic conductivity, the analytical model predicted leachate volumes and flow satisfactorily against measured values in a laboratory column. While these outcomes may not represent field landfill experience it informs confidence levels around this kind of representation for landfill simulation.

More sophisticated mathematical models have also been used in research to evaluate gas and heat generation and transport processes in landfills. Prediction of gas production rely on mathematical models simulating various stages of anaerobic degradation, typically by methodologies such mass balance, zero-order, first order, second order rate equations or a combination of these (Sel, Çakmakcı et al., 2016). To simplify and better estimate landfill gas (LFG) production capacity, Majdinasab, Zhang et al. (2017) reviewed previous methane estimating mathematical models and concluded that while higher order models had less inaccuracies, their accompanied higher complexity make first order decay models preferable for users. They also hold that Monod type first order kinetic decay models represent the majority available and that multiphase approach models are used to a lesser extent. Like Lu, Eichenberger et al. (1984) they also found most equations for gas generation were of the type $XYZe^{-kt}$ but they generalised XYZ as various values considered in the mathematical models depending on factors such as landfill conditions (temperature and climate), waste quantity and waste quality (degradation over time, carbon content, moisture and age of waste).

It is important to note that while mathematical models are useful, the diversity and heterogeneity of waste and microbes involved in degradation make accurate predictions impossible. Mathematical models provide an opportunity to predict landfill traits, helpful in experimental validation, landfill design, operation, and control.

2.8.   Geotechnical Properties

2.8.1. Measurement of Geotechnical Properties

Most geotechnical properties of solid waste are based on standards for soils established by ASTM with incidental modifications to accommodate the material being tested. The understanding of leachate flow and transport mechanisms rely on soil science but the underlying criteria on which those transport processes are based are often not proved for waste. This difficulty arises because waste is more heterogeneous than most soils, and changes character over a time frame much faster than the geological timeframes experienced by soils. For example, Powrie and Beaven (1999) investigated the variations in density with vertical stress and identified that particle density increased with increasing stress. They maintain that as the nature of waste contains deformable and crushable particles, this is not surprising, however it may cast doubt on the applicability of conventional soil mechanics as it is normally assumed that the solid particles are incompressible.

Conceptual representations of geotechnical properties exist to describe stabilisation mostly designed for consideration of mechanical strength of waste for stability, well design or prediction of sidewall failure. As the experiment proposed herein relies on compression of waste in lieu of depth, these representations and models may be useful for the understanding and estimation of compression effects.

The compressibility of waste has been shown to relate to the degree of degradation (Hossain, Gabr et al., 2003). Sarsby (2013) also outlines various methods that relate compression to stabilisation and soil strength.

2.8.2. Compression

Assessing mechanical strength of waste using geophysical and biogeochemical analysis is important for landfill design, prediction of final volumes, stability analysis and assessing risk of structural failure. These analyses are also useful for the understanding and estimation of compression effects. Waste is a highly compressible material yet magnitude and rates of compression are difficult to predict quantitatively (Powrie, Richards et al., 1998). Pulat and Yukselen-Aksoy (2013) summarise the characteristics of waste compaction as dependent on

- Water content
- Waste structure (anisotropy)
- Degree of degradation
- Particle size distribution
- Layer thickness
- Waste composition.

Compression can be considered to occur mechanically, because of degradation, ravelling and due to wetting. Figure 8 outlines these mechanisms of compression.
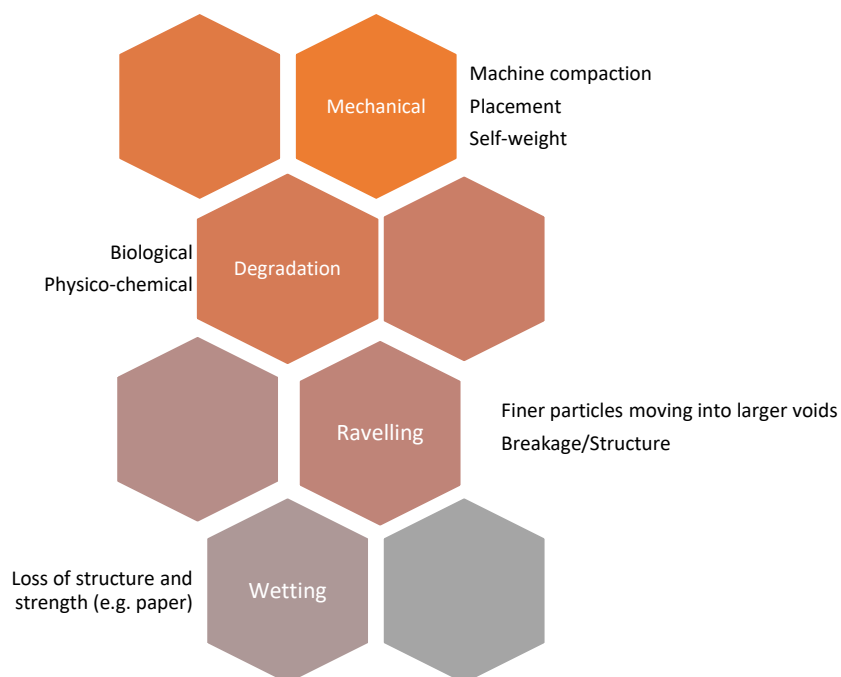
Figure 8 Mechanisms of compression

Gao, Bian et al. (2017) note that there are immediate (primary) compression mechanisms (e.g. machine compaction) and time dependent ones, such as biocompression and mechanical creep.

Mechanical compaction occurs via machine compaction at collection and waste placement and due to self-weight. Compaction during landfilling introduces immediate compression to the waste.

Zekkos, Bray et al. (2006) report from other studies unit weight values from 10 to 13 kN/m^3 (1,020-1,325 kg/m^3) near the surface at six California landfills, increasing to 13–16 kN/m^3 (1,325-1,632 kg/m^3) at a depth of 30 m. They further support other research stating that landfills with leachate recirculation are 'likely to be significantly higher than conventional MSW landfills, with values sometimes approaching or exceeding 20 kN/ m^3 [2,040 kg/m^3] at depth.' Old landfills have been known to settle 30% which further supports the range of these values.

Self-weight and biodegradation are also known secondary compression components. There is considerable uncertainty in the simulation of waste properties caused by secondary compression. The magnitude of secondary compression is due to similar factors discussed by Pulat and Yukselen-Aksoy (2013) above such as waste composition, water content, and depth of waste, as well as hydraulic factors that alter at depth such as permeability and pore properties. Hossain, Gabr et al. (2003) also highlight that the compressibility of waste is related to the degree of degradation. Portelinha, Correia et al. (2020) define biocompression, based on previous research, as mainly associated with anaerobic decomposition of the waste biodegradable organic fraction, which depends on MSW organic content, moisture content, and temperature.

2.8.3. Hydraulic Conductivity

Hydraulic conductivity is the ease with which a particular fluid flows through a particular material. Usually it is the ease of water through soil. It is dependent on the material; particle size distribution, roughness, tortuosity, shape and degree of interconnection of water-conducting pores; as well as the viscosity and density of the fluid (and therefore temperature). Considering the material characteristics and the heterogeneity of waste it is not surprising that the hydraulic conductivity for waste has a wide range varying between 10^{-3}-10^{-6} m/s.

Zeiss and Uguccioni (1997) showed that hydraulic conductivity and drainable porosity decreases with depth, as expected. Unpredictably, the standard deviation of moisture content increases with depth, they postulate due to channelled flow. In contrast, Durmusoglu, Sanchez et al. (2006) studied permeability during compression and found that variation in the coefficient of permeability did not follow any trend in relation to hydraulic gradient or moisture content. This is not surprising given the heterogeneity of waste. In large-scale tests, a few pieces of impervious material included in specimens could greatly impact the measured permeability.

Powrie and Beaven (1999) also investigated the variations in drainable porosity and hydraulic conductivity (as well as density) with vertical stress. They found, over the range of stresses applied, that hydraulic conductivity changed by a magnitude of three (3.4x10^{-5} to 3.7x10^{-8} m/s). Their results were able to be redistributed to various equations relating hydraulic conductivity to an overall depth of a landfill via vertical effective stress. This allowed a basis for measuring hydraulic conductivity within the experiment as an estimate of depth effects. Daniel (1993) cautions against relying on small scale laboratory hydraulic conductivity test results stating that these tests do not always give representative in-situ values.

Other researchers have experienced wide hydraulic conductivity ranges so it is likely that the outcome of hydraulic conductivity tests will be both variable and difficult to attribute to any one specific characteristic.

Waste is a large particle sand but as it degrades is more like a fine sand/silt and hydraulic conductivity can decrease by an order of magnitude. It is necessary to also consider the covering soil as a component of landfills. In Kate Valley landfill and the samples used particularly in this study, soil cover is a significant portion.

Table 2 outlines some characteristic geotechnical property ranges for MSW. Note these values are indicative, as reported values vary due to the differing composition, waste age and moisture content of samples. They also vary by location and over time within waste. The data is presented to allow understanding of the magnitudes of variation rather than actual values.

Table 2 Summary of waste geotechnical property ranges
Sources: Christensen, Cossu et al. (1989), Daniel (1993), Qasim and Chiang (1994), Qian, Koerner et al. (2002), Esteban-Altabella, Colomer-Mendoza et al. (2017), ISWA (2019)

Property	Range
Hydraulic conductivity	10^{-3}-10^{-5} m/s
MSW moisture prior to landfilling	10-25%
Landfill moisture content	25-35%*
Household waste	150 kg/m^3
Compactor truck	180-415 kg/m^3
Loose landfill	Up to 390 kg/m^3
Dozer compacted	475-593 kg/m^3
Compactor on landfill	534-1068 kg/m^3
Old waste at base of landfill	Up to 1286 kg/m^3

3. RESEARCH DESIGN

3.1. Research Objectives

The primary goal of this research was to begin an understanding of using different leachate removal depths for recirculation in landfills and in the process gain some knowledge about aged (10-year old) waste from Kate Valley.

The overarching premise for this proposal was that recirculating leachate captured prior to passing through the entire depth of a landfill will enhance anaerobic biodegradation and generate a better quality leachate than standard recirculating processes. What "enhancing anaerobic biodegradation" and "better quality leachate" means was to some extent answered in existing literature. These qualitative statements require quantitative assessment to ensure any conclusions are based on objective fact: specifically measuring chemical and geophysical properties which define degradation and stabilisation. The experiment needed to produce these types of results to allow supportive arguments based on existing research.

The central question to be answered is:

> "Can any benefit be measured in using leachate for recirculation from shallower depths compared to the bottom of a deep landfill?"

To answer this question the proposed experiment aimed to:

- Allow for leachate to be intercepted during its progression through the waste column
- Provide waste at different compactions to mimic degradation and hydraulic parameters at different depths
- Measure and examine gas and leachate parameters during the experiment
- Analyse waste pre- and post-experiment to support gas and leachate data
- Determine total methane potential to support outcomes.

The objectives were to:

- Gain some knowledge about the impacts of different leachate removal depths in landfills for use in recirculation.
- Outline possible models and concepts that will help assess the benefits of using intercept leachate
- Develop and run tests that will allow valid comparison of intercept leachate recirculation against bottom leachate recirculation
- Examine the results of the experiment to quantify benefits (if any)
- Examine previous research and interpret it against the results to qualify, understand and explain the results achieved
- Provide some useful information about the state of 10-year old waste at Kate Valley Landfill.

Figure 9 details associations between relevant questions and where or how, the answers may be sought based on existing knowledge.
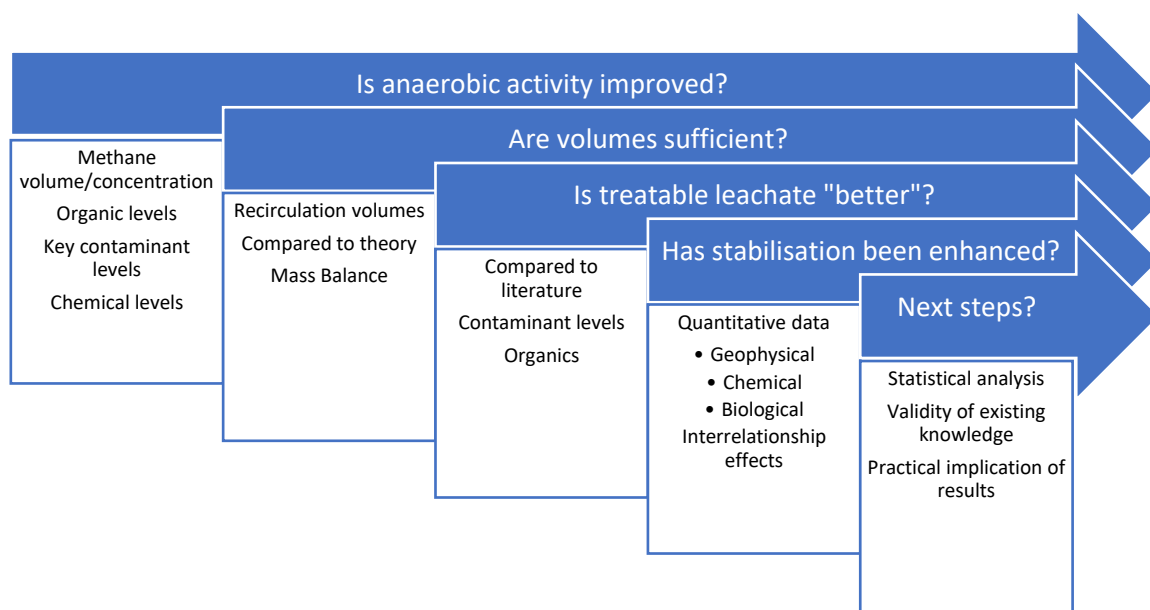
Figure 9 Cascading research questions

There may not be definitive answers for these questions, but this initial research will advance and help develop the knowledge needed to effectively use leachate recirculation in NZ landfills, especially given the lack of information found to date. While no studies using intercept leachate have been found, it did not mean studies have not been done. The lack of information indicated that consideration is rare and raises an opportunity to discuss the concepts.

This work was intended to be an evaluation of concept and does not address practical issues for implementation in a full-scale landfill.

3.2.   Methodology

Reddy, Kumar et al. (2017) asserted that the test setup and scale of the experiments conducted on waste have a profound effect on waste behaviour. They maintained it was difficult to decide on size and general test setup of laboratory apparatus to accurately simulate the behaviour of highly heterogeneous and anisotropic waste in landfills. Both Reddy, Kumar et al. (2017) and Gabr and Valero (1995) argued that field heterogeneity and anisotropy were often not well represented in laboratory experiments however, they recognised that to obtain a thorough understanding of an attribute, and therefore whether or not any of its parameters were significant, a reductionist laboratory approach is required. Fei and Zekkos (2013) monitored the degradation process closely in mesoscale laboratory simulators and noted their value in isolating the contribution of each settlement mechanism. The effects of initial and operational conditions can be better understood because they can be better controlled in a laboratory environment.

To begin this investigation we need to learn the differences between using bottom and intercept leachate for recirculating in hastening stabilisation. The relevant parameters to indicate settlement, stabilisation and degradation of waste include the following aspects:

- Leachate quality
- Gas volumes
- Gas composition
- Waste characteristics
- Settlement indication
- Applicability of theoretical models.

The experiment was designed to achieve the objective of facilitating comparison with other research and to enable intercept leachate to be easily collected or to allow the leachate to continue to pass through the waste. To meet these criteria the experiment simulated the top and bottom of a landfill by having two separate containers (test cells) with low and high density waste. This reflects surface waste that has been compacted only via primary compaction (low density) and waste nominally 30 m deep reflecting secondary compaction (high density). Estimates of density were discussed in section 2.8.

Combined these two test cells resulted in a test unit that helped simulate the travel path of leachate in a landfill and achieved the objective of ensuring valid comparisons between intercept and standard leachate recirculation.

Verification that there was consistency within and between test cells and between samples taken for analysis was an important factor in the experimental design.

The individual test cells needed to be big enough to accept raw waste – to avoid the added complexity of waste samples not representing the landfill – and have a consistent heterogeneity between and inside each test cell. It needed to be small enough to mitigate safety issues around gas volumes and physically moving the waste within a laboratory environment. There were also restrictions regarding space available given the need to be in a temperature controlled environment, and sharing this space with other researchers. Table 3 outlines the primary factors, requirements and risks considered when designing the scale of the experiment.

Table 3 Balancing experimental requirements.

Consideration	Requirements	Risks
Test cell dimensions	Consistency between test cells Alignment to other research Lab space available Adequate headspace and drainage layers	Unsafe environment or poor practices if dimensions are too large Inadequate results or inability to reconcile against other research Poor leachate flow, impaired drainage Insufficient headspace for gas collection
Waste volumes	Sufficient for anaerobic activity Sufficient gas volumes for analysis Restrictions in laboratory space Manual handling considerations	Handling the required volumes of waste Losing heterogeneity of waste Insufficient volumes to sustain microbial activity
Waste selection	Already in advanced methanogenic stage Consistent samples Confidence in consistency	Expose waste to oxygen Release previously inaccessible organics for pre-methanogenic bacteria activity Sample contamination
Waste sizing	Consistency in sample Consistency within test cells Consistency between test cells Alignment to other research Alignment to in-situ waste properties	Sorting waste removes heterogeneity of waste that occurs in landfills Sorting waste changes geotech properties Large particles allow leachate by-pass
Leachate recirculation	Support microbial activity Sufficient leachate flow Sufficient leachate volume Access and volume excess for sampling Consistency between test cells	Inconsistencies due to heterogeneity Restrictions in laboratory space Insufficient leachate flow Slow leachate flow
Gas production	Sufficient volumes to measure degradation Safety of gas volumes in confined atmosphere	Restrictions in laboratory space High volumes No gas production Ensuring gas can be removed from test cell and is vented
Microbial environment	Temperature sufficient to support microbes Ensuring no exposure to oxygen	Maintaining equivalent conditions to in-situ landfill waste Heterogeneity of waste that occurs in landfills Insufficient quantity of waste Exposure to oxygen kills anaerobic microbes

Changes to chemistry, biological activity and waste settlement were expected to occur and were measured. Table 4 outlines tests expected to assist in interpreting the experimental outcomes. Testing is to applicable standards where available. For example, ASTM D2974 – Standard Test Methods for Determining the Water (Moisture) Content, Ash Content, and Organic Material of Peat and Other Organic Soils.

While the suite of analytes for soil and groundwater contamination is quite well standardised, there is variety in analyte selection used in studies of landfill leachate. Chemical and biological analyte selection in previous research guided the selection in this study. Gettinby, Sarsby et al. (1996)

summarise a number of previous studies and this was used as a starting point for selection. Similarly Gabr and Valero (1995) outline many of the possible geotechnical tests available.

Table 4 Proposed experimental testing parameters.

Parameter	Test frequency	Reason
Waste		
Hydraulic conductivity	Initial & final	Context, validation
Density	Initial & final	Context, validation
Particle size	Initial & final	Context and characterisation
Permeability	Initial & final	Context, characterisation & comparison
Water content	Initial & final	Context and characterisation
Organic content	Initial & final	Context, characterisation & comparison
Leachate		
Organic content (COD/BOD/TOC)	Monthly	Context, characterisation & comparison
Volatile fatty acids	Monthly	Confirming degradation
Total Nitrogen	Initial & final	Comparison with other research
Ammonia	Initial & final	Comparison with other research
Chloride	Initial & final	Characterisation & comparison. Interference
Phosphate	Initial & final	Characterisation & comparison
Sulphate	Initial & final	Interference
Iron	Initial & final	Interference
pH	Regularly	Characterisation & comparison
Gas		
Methane concentration	Daily	Characterisation & comparison
Methane flow/volume	Continuous	Characterisation & comparison
Carbon Dioxide concentration	Daily	Characterisation & comparison
Hydrogen Sulphide	Daily	Interference
Oxygen	Daily	Leaks, test cell issues

It was also intended to conduct a post-mortem on the waste samples for analysis at the end of the study. It was hoped that a quantitative analysis of the two densities of waste post experiment in terms of remaining organic content, hydraulic conductivity and settlement would support other data.

The remainder of this section details these considerations concluding with a description of the experiment as built and the equipment required.

3.3. Test Cell Design

3.3.1. Test Cell Dimensions

Fei, Zekkos et al. (2014) presented a summary of laboratory simulations and noted that testing objectives were related to configurations but not dimensions. They further maintain that for biodegradation tests of waste, the size of the simulator was important. They cited a minimum diameter of 300 mm for use of unshredded waste to reduce bias in results, especially during geotechnical testing, and this reduces preferential flow paths in the horizontal direction. This sets a minimum diameter of 300 mm for the test cells used in this study.

The easiest test cells to use was an existing container such as a barrel or drum. The use of a cylinder shape minimises surface area to volume, reducing the potential for preferential pathways against the walls and helps retain heat.

A minimum diameter of 300 mm for the test cell allows a maximum size of waste to be set at 30 mm This allows for sizing by hand in a pragmatic manner during loading. Selection of a height close to the diameter also reduces the risk of a large item blocking a thin cylinder or forming a short circuit down a wide cylinder.

A minimum of 50 mm headspace for gas and 100 mm bottom for drainage aligns the experiment with most other laboratory and mesoscale experiments; Pohland (1975), Christensen, Cossu et al. (1989), Leuschner (1989), Stegmann and Spendlin (1989), Qasim and Chiang (1994), Senior (1995), Reinhart and Townsend (1998), Fei and Zekkos (2013), Sivakumar (2013), Reddy, Kumar et al. (2017), Bareither, Wolfe et al. (2013).

The experiment was designed to simulate a landfill and incorporate two test cells vertically aligned making up a test unit. Appendix C1 – Test Cell Dimensions describes the space restrictions and configuration further. This allowed a decision to utilise a 60 litre (nominal) polyethylene off the shelf barrel as the test cell. The amount of waste was initially estimated at 400 litres or 320 kg, plus an additional 25% for testing and adequate sampling. Thus 400 kg of waste was sampled and removed from Kate Valley.

Qian, Koerner et al. (2002) outlined requirements for drainage layers; specifically that they required:

- High hydraulic conductivity
- Good stability
- Resistance to plugging or clogging
- To serve as a filter to retain overlying waste particles and stop them migrating down
- To ensure the hydraulic conductivity of the waste is much less than the drainage layer.

Sand, gravel and filter cloth or geotextile in combination is usually utilised to achieve this and were incorporated in this study's test cells.

3.3.2. Waste Selection

As raw waste has a 12 month lead time (Qasim and Chiang, 1994) before degradation takes place, access to partially degraded waste samples and active leachate from a NZ landfill was required to ensure anaerobic processes relevant to waste were underway and can continue. Kate Valley Landfill agreed to supply this waste and had confidence in their records to be able to source waste that was 10-years old. Figure 10 shows that the waste was at the age expected. Waste at this age was expected to be in methanogenic state, especially given that Kate Valley produced landfill gas in sufficient quantity to produce electricity at the end of 2014, 9½ years after receiving their first

waste. There was a balance that needed to be achieved with the samples to ensure consistency across the test cells. The waste needed to be consistent across the eight test cells (four test units in duplicate) yet heterogeneous within each test cell. A minimum amount of disturbance to the waste was intended (to avoid disturbing the microbial communities) but large items needed to be removed to ensure they did not block flow paths in the test cell.

Figure 10 Proof of waste age: a 2010 brochure

The waste collected from Kate Valley was dug from a trench in an area the operators knew the waste was 10-years old. The trench was dug as deep as the excavator arm could reach and dumped on the ground. Samples were quickly transferred into 20 litre buckets with waste selected from more than one area to ensure a mixture of waste. A total of 480 kg was taken to ensure enough was available for both the test cells and to determine waste characteristics. Large pieces of waste were preferentially removed as some

discarded waste items were metres in dimensions. Figure 11 illustrates relative sizes of the waste particles on-site; for example offcuts form industrial processes were up to 4 m x 3 m. Other examples include lengths of timber, branches, entire full garbage bags and plastic wrap. Pieces larger than 50 mm in any dimension were not transferred from the landfill. The quantity of material

Figure 11 10-year old waste excavated ready for sampling

excluded is estimated at 25% by weight. This will have affected properties of the waste such as density, hydraulic conductivity and compactability, resulting in leachate and gas flows different to Kate Valley landfill. However, there was a balance to be struck between experimental dimensions and waste sample representation.

The waste samples selected for the test cells were generated from a minimum of four buckets to again ensure a mixture of waste and consistency between test cells. It was observed during transfer of the waste from landfill to buckets and bucket to test cell that the waste had different

colours, textures and especially plasticity (clay) content. The waste was moist rather than wet (as described by ASTM D 2488), free liquid was not apparent. Fine soil-like components did not adhere to the spade used unless large clay particles were encountered and the dry strength was medium with most particles breaking with considerable finger pressure. After loading, the test cells were sealed and purged with nitrogen to displace oxygen from the system.

3.3.3. Leachate Recirculation

Acquiring a target amount of moisture to add was a necessary step in designing and operating landfills as well as for simulation experiments. The objective of adding moisture in this experiment was to hasten stabilisation. Waste stabilisation was likely to be optimal at near saturation conditions where liquids were moving quite rapidly. In operating landfills it is not feasible nor safe to run at this level of moisture, therefore moisture content was targeted at near field capacity. In an ideal landfill all moisture would be absorbed by the waste until field capacity is reached. Pragmatically the fundamental processes governing fluid flow in porous media means it is impossible to bring all waste to field capacity without some of the waste achieving saturation, and then draining by gravity. This means while field capacity was a target, it was not an outcome for success.

Tolaymat, Kim et al. (2013) state field capacity was a reasonable target, but outline the complexities in determining moisture requirements including:

- Varying stoichiometric requirements
- Assessment of initial moisture content
- Waste composition
- Waste density
- Heterogeneity
- Effects of compaction

Tolaymat, Kim et al. (2013) also indicated that landfills struggle to introduce enough moisture to reach the target moisture content (i.e. field capacity) and the ability to cycle liquids through the landfill multiple times was often not a realistic consideration. It was anticipated that this may be an issue in this study, therefore leachate was collected from Kate Valley landfill to obtain the field capacity target and to use as an additional leachate source if recirculation alone was inadequate once field capacity was reached.

To design the experiment prior to waste sample collection, leachate volumes were estimated using data from previously published research. Appendix C2 – Field Capacity, Appendix C3 – Moisture Addition, and Appendix C4 – Leachate Recirculation details these assumptions and calculations.

3.3.4. Compaction

Compaction of the waste in the test cells was used to alter the density and permeability characteristics of the waste, and to mimic flow restrictions in a landfill as a consequence of depth. The waste samples were compacted to simulate waste density at two different depths within a landfill. The densification of waste in this experiment was in effect primary compression; it

represented density levels at the surface and at 30 metres depth. This would include the time-dependent factors as outlined by Gao, Bian et al. (2017) (see Section 2.5).

Various mathematical models and methods for predicting stabilisation and soil strength reviewed by Sarsby (2013) gives a starting point for considering physical properties such as density and permeability of the experiment.

To achieve the densities required in the test cells, hand-tamping was used to compress raw waste collected. Appendix C5 – Compaction outlines the outcomes to find a method of compaction to achieve the densities required to simulate a landfill within the test unit.

3.3.5. Gas Production

One of the greatest risks in the experimental work was the production of methane. Methane is flammable gas and potentially explosive, with Lower Explosive Limit of 4.4% by volume and an Upper Explosive Limit of 15%. Civil and Natural Resources Engineering (CNRE) environmental laboratory staff work with anaerobic digestion continually and have experience, protocols and procedures to ensure the risks around methane were mitigated. However, restricting volumes was still an important consideration. There was a balance between having an experiment large enough to produce gas for results but at the minimum possible methane volumes to mitigate risks. For waste decomposition, this translates back to waste volumes. Qian, Koerner et al. (2002) for instance outlined landfill gas total volumes as a direct function of the total quantity of organic material in the waste available for decomposition.

To assist in the dimensioning of the experiment it was necessary to estimate the potential volume of methane and total gas produced. This was achieved by assessing gas production rates from either landfills or simulated landfill experiments conducted by previous researchers. This analysis was intended to assist in the validity of various mathematical models once gas volumes have been obtained from the experiment.

Previous studies and summary reviews from the literature can be used to determine likely levels of gas production, although Majdinasab, Zhang et al. (2017) point out there was considerable uncertainty in obtaining generation rates due to experimental parameters having significant uncertainty themselves (including temporal and spatial heterogeneity). Table 5 summarises published gas production outputs from previous researchers, adapted where possible to reflect 10-year old waste. It should be noted that much research surrounding methane potential does not present data in rate format (volume per mass rather than volume over time). While assumptions around time for methane to be exhausted can be assumed, this adds another layer of uncertainty and was not included unless the research specifically declared time-frames.

Table 5 Published gas production rates from different researchers

Reference	Source of data	Maximum Basis as quoted	Calculation Assumptions	Rate (l CH_4/kg waste/day)
Pohland (1980)	Georgia Institute of Technology Study	0.032 m^3/dry kg/yr @ 57% CH_4	25% moisture	0.037
Ham and Barlaz (1989)	Lysimeters	0.001-30 l/kg/yr total gas	70% methane	0.058
Ham and Barlaz (1989)	Pilot scale	15-60 l/kg/yr total gas	70% methane	0.115
Ham and Barlaz (1989)	Field studies	10-20 l/kg/yr total gas	70% methane	0.038
Milke (1992)	Previous literature	10 l CH_4/kg dry/yr, 300 l/kg dry at 20 years	70% methane 25% moisture	0.014 0.022
Pohland, Cross et al. (1993)	USEPA report based on 10 simulated landfill columns	378 kg of shredded waste^ 42 m^3 gas over 3 years		0.101^
McBean, Rovers et al. (1995) from Ham and Barlaz (1989)	Theoretical	Theoretical 520 l/kg total* By degradability 300 l/kg Projected from data 400 l/kg	20 years	0.071# 0.041 0.055
Giardi (1997)	Landfills	20.76 m^3/t/yr (LFG)	70% methane	0.040
Reinhart and Townsend (1998)	Breitenau Landfill, Austria test cells	5.31 x10^{-6} m^3/d/kg MSW gas @58.4% CH_4		0.0031
Reinhart and Townsend (1998)	University of Florida in Orlando recirculating landfill	0.0236 m^3/kg-yr		0.065
Reinhart and Townsend (1998)	Brogborough, UK, pilot cells	5.5-11 m^3/tonne/yr (theory) p45	70% methane	0.021#
Qian, Koerner et al. (2002)	Empirical data of landfills	6.24 m^3 (LFG)/Mg (waste)/yr	70% methane	0.012
Majdinasab, Zhang et al. (2017)	ADEME model	100 m^3/Mg^{-1} in 80 years	70% methane	0.0024

* McBean, Rovers et al. (1995) show a minimum of 30% at year 5 to a maximum of 90% at year 10. Our waste was within this range (10 years)

Theoretical or ideal values expected to be higher than test cells

^ Expected to be high as waste was shredded and therefore more accessible to microbes

Qian, Koerner et al. (2002) supply a suitable estimation of gas generation rates using a cumulative calculation based on waste age and mass. The equation used is:

$$(Q_i)_t = 2kL_0M_ie^{-kt_i}$$
(3)

Where:

Q_t = expected gas generation rate for waste mass, M_i, (m^3/yr),
k = methane generation constant (yr^{-1}), (suggested range of 0.05-0.15 yr^{-1}),
L_o = methane generation potential (m^3/tonne), (suggested range of 140-180 m^3/tonne),
M_i = mass of solid waste (tonne), estimated for this experiment as 346.8 kg total,
t_i = age of the waste mass, Mi (yr), known to be 10 years.

For safety reasons, maximum methane levels were estimated to give a volume rate of gas for the 10-year old waste used in this study of 0.09 l/kg waste/day. Comparing this to other literature values given in Table 5 of between 0.018 and 0.11 litres CH_4/kg (dry waste)/day allows the estimated level of 0.09 l/kg dry waste per day maximum to be accepted as a reasonable design value. These figures give rise to a maximum volume of methane levels expected as 35 litres total per day. As the factors used to calculate this estimate were conservative this volume can be assumed to be a maximum.

3.3.6. Microbial Environment

Control of the temperature of the test cells was limited to running the experiment in a temperature controlled room (set at 38 ºC). This was inside the optimum temperature range for methanogenesis (Senior, 1995).

Oxygen on the other hand is detrimental to anaerobic bacteria. It was important to ensure the waste was not exposed to oxygen especially during the experiment, to maximise survival of the methanogenic bacteria. The test cells were purged with nitrogen and sealed to exclude oxygen and to support anaerobic activity under mesophilic temperature. It was noted the test cells bulged a little when the pressurised nitrogen purge occurred indicating a good seal was present. At no stage during the experiment was gas able to be smelled giving another good indication that leaks were not occurring.

The test cells were made of HDPE polymer and therefore have the potential for oxygen gas to permeate through the walls. Estimation of oxygen transmission rate using HDPE supplier technical information (EvalAmericas, 2021) gives a maximum oxygen transmission rate of 3.2cc/24hr or less than 0.02% each day (see Appendix C7 – Oxygen Transmission Rates Through HDPE). This is less than the error of the gas analyser during measurement. Rising oxygen levels would be a sign of possible permeation issues and it is recommended that this estimate be revisited if oxygen ingress is suspected.

3.3.7. Test Cell Arrangement

A large bench-scale system was set up in the Environmental Laboratory in Civil and Natural Resources Engineering at the University of Canterbury. A schematic is shown in Figure 12 and the final physical setup as Figure 20.
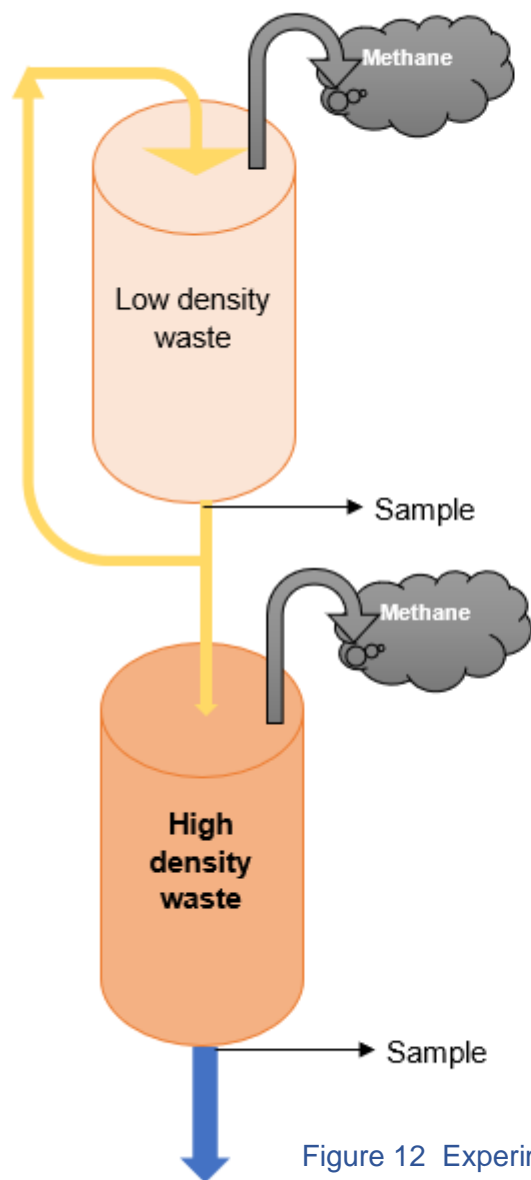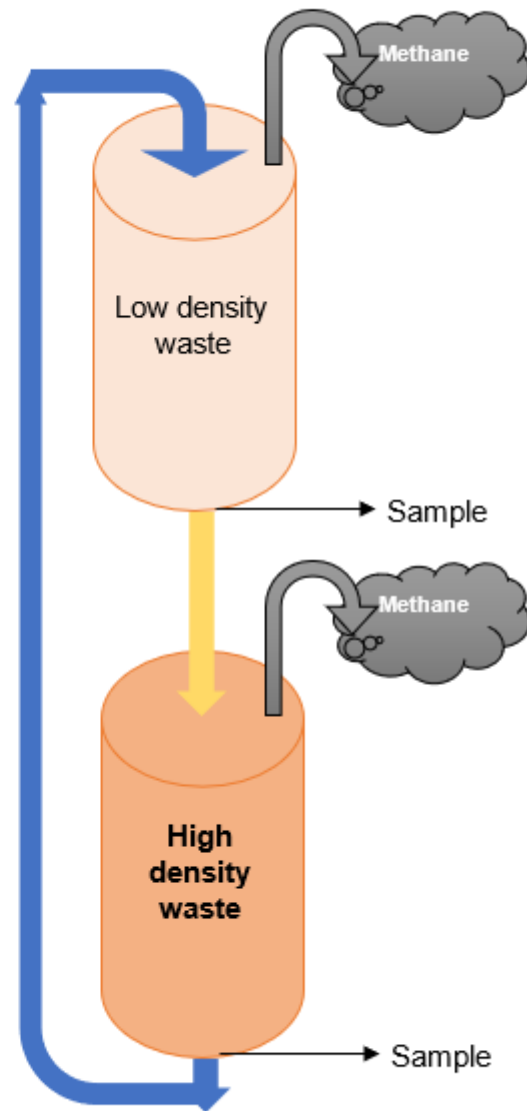
Figure 12 Experimental setup

The system consisted of test cells (made from 60 litre barrels) filled with compacted 10-year old waste to simulate landfill hydrology and to explore leachate hydraulic and quality characteristics.

Two test cells, one on top of the other formed a test unit and are intended to represent the top and bottom half of a landfill. The top test cell had lightly compressed waste, simulating waste at the top of a landfill and the bottom had more highly compacted waste to simulate the density of waste at the bottom of a landfill. This system was replicated to allow one set to run with leachate recirculated after it had travelled through both halves (right hand side Figure 12) and one had the leachate intercepted mid-way between the top and bottom test cells (left hand side Figure 12) for recirculation.
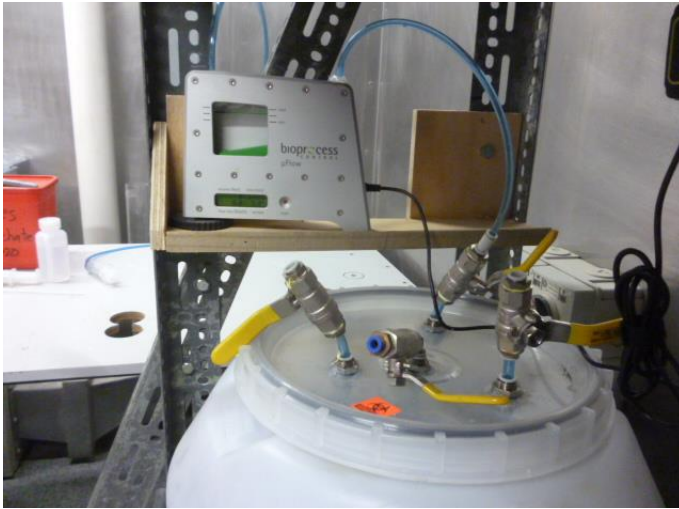
Figure 13 Top of a test cell showing valves and gas flow meter

Leachate recirculation consisted of removal of leachate via spigot just off the base of the barrel and 360⁰ garden micro-sprayers to disperse leachate over the top of the waste via either a peristatic pump or syringe. The lid of the test cell contained three valves. One to feed the microsprayer to allow the leachate to be recirculated (manually) and sprinkled on the waste, one for gas volume measurement and release, and two others for intake and return for the gas analyser. Figure 13 shows these valves.

Leachate volumes were measured and leachate was sampled for quality characteristics and returned to the cells, as appropriate. Gas volume and composition were regularly measured.

The standard leachate recirculation test unit allowed flow from the top test cell to be passed to the bottom test cell (less any sampling required) and from the bottom test cell back to the top test cell (less any sampling required). Leachate was topped up in the low density test cell initially to get waste to field capacity, with additional leachate added to balance sample volumes taken. Figure 14 depicts this flow regime.
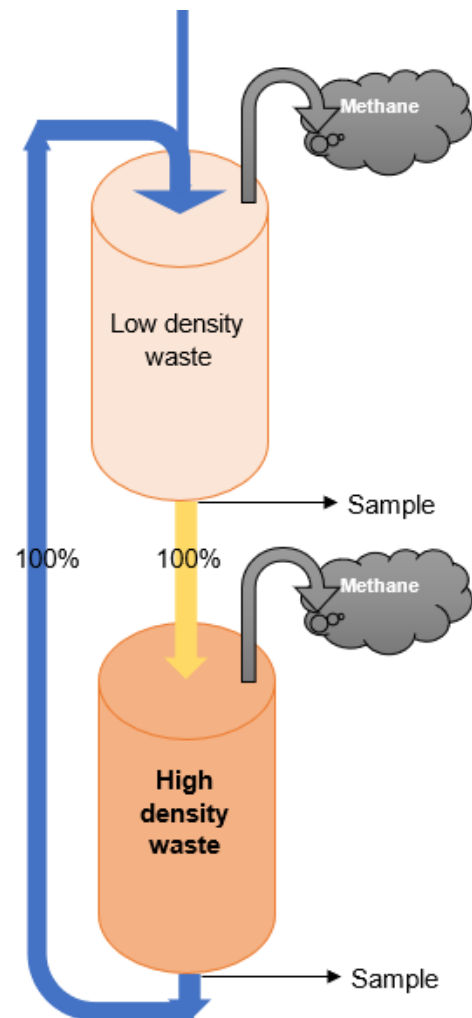
Figure 14 Standard leachate recirculation flows

There was always going to be compromise necessary to recirculate intercept leachate without knowing hydraulic conductivity and flow times or the leachate characteristics over time. The basis of all decisions in how to structure the experiment was to eliminate as many non-core variables as possible. The first decision in designing the intercept leachate recirculation test unit was what to do with the leachate remaining once it had passed through the entire test unit. To eliminate any possibility of killing the methanogenic bacteria through build up of toxins or by slower flow rates in the high-density waste test cell it was decided to dispose of it. It was further decided to merely split the flow in half allowing half into the bottom test cell and recirculating half back into the top test cell. This sets up the requirement for additional leachate to maintain flow. Therefore any additional fluid requirements consisted of the raw leachate collected from Kate Valley. Nominal percentage flows are shown in Figure 15. Additional leachate was available to top up both the low density and high density test cells to get waste to field capacity, ensure sufficient flow through the top test cell and to top up sample volumes taken.
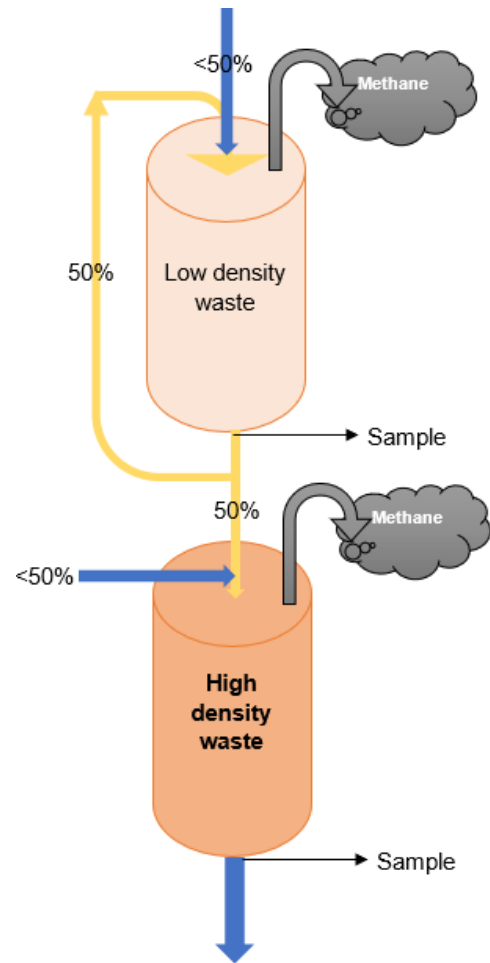
Figure 15 Intercept leachate recirculation flows

Table 6 outlines the main resources used for this study.

The basis for these parameters appear in Table 4 and methods applied for analysis appear in Section 4.

Table 6 Resources required for experiment

Resource	Basis of need
Eight (8) test cells. Each test cell is identical and nominally 60 litres	Refer to Figure 12 and Figure 20. Duplication for confidence and/or redundancy.
Piping, valves fittings for test cells	To allow for manual leachate recirculation, gas collection and analysis and sampling.
Framework to support test cells	Access, stability, safety (Figure 20).
Methane flow meter	Gas measurement and safety.
Geotech laboratory resources	Testing of waste characteristics.
Environment laboratory resources	Testing of chemical and biological waste, leachate and gas properties.
External laboratory resources	Testing of chemical and biological waste and leachate properties unable to be done in the CNRE laboratory.
Landfill meter	Gas analysis, specifically methane and carbon dioxide.
pH meter	Measuring pH
Measuring cylinders	Leachate volumes removed
Peristatic pump	Leachate return
Syringe (60 ml)	Leachate return

3.4.   Biomethanation Potential via Tube Anaerobic Respirometer

A simple technique using a tube batch reactor (Figure 16) to assess biomethanation potential of organic waste was developed and engineered by the environmental Laboratory in the CNRE department at the University of Canterbury, with further research carried out by Dr Nastein Qamaruz-Zaman (Zaman, 2010). The tube batch reactor is an anaerobic respirometer that uses digested sewage sludge (DSS) as seed. It has a wide mouth enabling analysis on larger sample sizes with less sample disruption or interruption than traditional methods. This method avoids the problem of laboratory scale experiments using small sample sizes which may not be representative of the bulk material. This is a particular issue for heterogenic solid waste with large particle sizes.

These tube reactors are known to be applicable to a range of substrates of varied degradability. The tube reactor is a 10 cm diameter PVC pipe, measuring 400 mm long with a 3400 ml capacity, capped at both ends with the bottom endcap fixed, and the top screw capped for easy sample loading. The top also contains valve penetrations for respiration and gas analysis. The valve used for gas analysis (labelled 1 in Figure 16) is connected to the exhaust port of the gas analyser. Inside the valve extends down a few centimetres towards the bottom of the headspace. This ensures gas is mixed and the sample drawn into the gas analyser cannot short circuit. This is especially important for methane, hydrogen and hydrogen sulphide species, which are significantly different in density to air and thus separation becomes possible.

1 →	1. Valve (remains closed until connected to gas analyser return)
2 →	2. Valve (remains open when connected to Tedlar bag) Also used for gas sample extraction during gas analysis
4 →	
3 →	3. Tedlar bag for gas collection
5 →	4. Screw cap
6 →	5. PVC pipe as reactor body
	6. Glued endcap
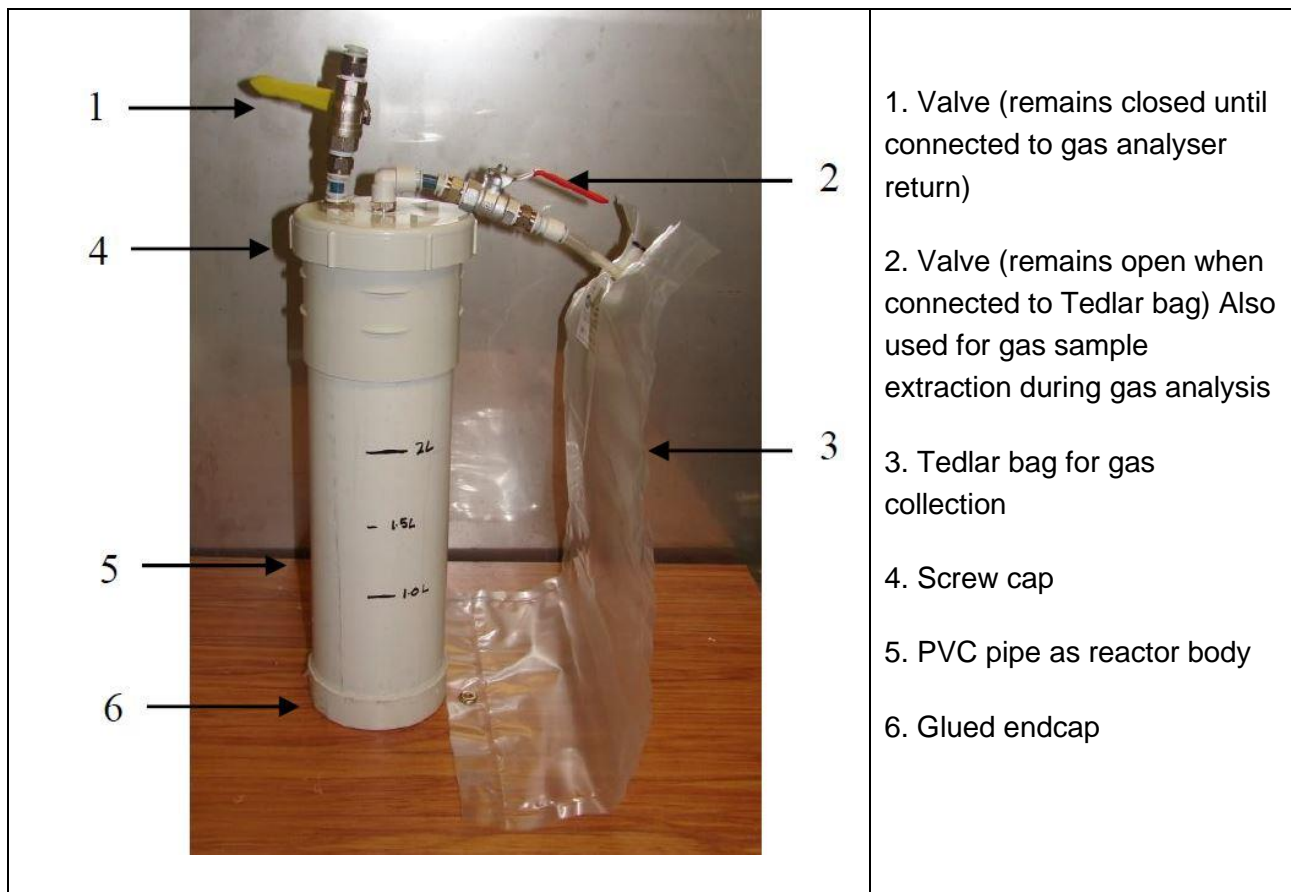
Figure 16 Tube batch reactor or anaerobic respirometer.
Adapted from Zaman (2010)

Zaman (2010) states the tube reactors operate best using pre-digested DSS as seed. The DSS is rested for five to seven days, with minimal mixing, without oxygen, at mesophilic temperature and without the addition of nutrients or buffer. This readapts the inoculum to the test temperature and helps to remove most of the remaining methane production of the seed, which could otherwise affect the test. Pre-digesting for longer is not advised as this can prolong the onset of subsequent methane production by the bacteria in the seed.

The tests planned included loading multiple tube reactors with the following sources of substrates:

- Blanks (DSS only)
- Rice
- Raw waste as collected for the test cells (stored for 6 months)
- Spent waste from intercept leachate recirculation test cells
- Spent waste from standard leachate recirculation test cells
- Raw waste resampled from Kate Valley at the time the tube reactors were loaded (called "refreshed waste", i.e. not stored for 6 months prior to use)
- Raw waste with rice
- Refreshed waste with rice.

This regime would allow the biomethanation potential of the raw waste to be assessed, allow inferences about any activity that may have occurred during test cell experimentation, ensure any

effects of raw waste storage are apparent, ensure biomethanation results recorded are related to the presence of bioavailability of organic material and not inhibited by other factors.

3.5.   Risk Analysis

The first risk identified was the risk of the experiment failing. Consideration was given to running another test unit without any leachate recirculation. The basis for this was as a contingency if the experiment failed. This option was rejected because the results would not be relevant to the basis of the hypothesis, so failure would require redefining the objectives rather than pursuing the reasons for failure. If failure occurred early enough the backup plan was to reset the experiment in-situ or obtain more waste and re-start. If this was not possible then the plan was to discover why the experiment failed and what could have been modified to ensure a successful experiment in the future.

There were multiple areas for uncertainty stemming from sampling, measurement limitations and human error. Sampling uncertainty was mitigated as much as practical and measured using repeatability. Measurement uncertainty was minimised using repeatability. Method precision and sensitivity were identified within the standard methods. Human error, when identified, resulted in repeating measurements, applying estimates or discarding results.

Statistical analysis had the potential to demonstrate that there was no significance in the results. This is a valid outcome indicating that there may well be no difference in leachate recirculation performance based on depth of leachate extraction. However the reductionist approach adopted in this study is an early stage of developing intercept leachate knowledge and is unlikely to be definitive.

Health and safety risks assessment around the experiment itself are outlined in APPENDIX B – Protocols. Health and safety issues around the field activity at Kate Valley are not included herein but exist with the University of Canterbury Field Activity Planning protocols.

4. METHODS

4.1. Experimental Measurements

It was assumed the extent of stabilisation could be evaluated principally by monitoring leachate and gas attributes over time. Waste characteristics and settlement (if any) within the test cells were also to be measured. This aligned with the literature review outlined in sections 2.3 through to section 2.5, section 2.7 and section 2.8. Table 7 outlines the types of variables that were monitored. Note this is a subset of those variables initially considered due to complexity or duration of applying a relevant method, resources available or expectation of meaningful results.

Table 7 Monitoring variables

Aspect	Shows	Expectation
Gas composition	Bioactivity	Enriched methane and carbon dioxide levels, lack of oxygen.
Gas volume	Bioactivity	LFG production over 180 days
Leachate organic content	Methane potential	High and reducing organic content
Leachate inorganic content	Leachate flushing	Varying depending on analyte
Leachate volume and flows	Waste characteristics	Once at field capacity balanced flows expected.
Waste composition	Methane potential	High volatile solids/organic content Alignment with literature
Waste size	Waste characteristics	Alignment with literature
Waste settlement	Waste characteristics Bioactivity	Alignment with mathematical models Not measured

Quantifying properties of waste materials is difficult, as described in section 2.2, because of its:

- heterogeneous nature
- high deformability
- temporal variation
- variable particle size
- problems with sampling under anaerobic conditions
- the need for large-size samples.

Measuring waste characteristics was still necessary to align with and understand differences with other research. The following section outlines those characteristics of waste desired to be measured.

4.2. Waste Characteristics

4.2.1. Composition

To determine the composition of the waste, samples independent of the test cells were screened at 10 mm and manually sorted. Since it was not possible to visually determine the composition of finer particles, the sample fraction greater than the 10 mm was sorted into several categories. These categories were selected as a compromise between the ability to select them out of the waste and the need to align categorisation with other research. Final selection of categories were:

- glass
- foam
- plastic (all kinds except foam))
- paper and cardboard
- wood
- textiles
- fibres (because of the age of the waste, ravelling was prevalent and it was often difficult to distinguish the type of fibre)
- metal
- stones
- clay or soil

4.2.2. Chemical Oxygen Demand

COD of the waste solids was by performed by Hills laboratory. Samples supplied were subsamples from different buckets of waste obtained and sieved to <2mm as required by the laboratory. The sample was air dried at 35ºC and the determination of COD was via dichromate/sulphuric acid digestion then colorimetry. Detection limit of 2,000 mg O_2/kg dry weight was reported.

4.2.3. Moisture Content

The moisture content was measured as the loss of weight after drying a sample to a constant value in an oven at 105°C. After cooling in a desiccator the moisture content was measured by weight loss from the raw waste sample and expressed as percent wet weight (ASTM, 1998). The expression for calculating moisture content on a wet basis is:

$$\% \text{ Moisture content} = (w - d)/w * 100$$

Where w = initial (wet) weight of sample, d = final (dry) weight of sample

4.2.4. Total Volatile Solids

Total volatile solids was based on Standard Methods for the Examination of Water and Wastewater: Standard Methods 2540G. Although Standard Test Methods for Determining the Water (Moisture) Content, Ash Content, and Organic Material of Peat and Other Organic Soils (D 2974-20) is often preferred in the agriculture, forestry, energy, horticulture, and geotechnical fields, standard procedure in the CNRE environmental laboratory is Standard Methods 2540G.

The method adopted was to firstly determine total solids. Total solids is the term applied to the material residue left after drying a sample in an oven at a 105 °C (as described for moisture content in section 4.2.3). Total solids included total suspended solids and total dissolved solids.

Specifically the method includes evaporating the sample in a dish and drying to constant weight in an oven at 105°C. The total solids in percentage of wet sample were calculated as:

$$\% \, Total \, solids = (C\text{-}A)/(B\text{-}A)*100 \qquad (4)$$

A = weight of dish (g)
B = weight of wet sample + dish (g)
C = weight of dried residue + dish (g)

The total solids were then heated to a constant weight at 550°C and cooled in a desiccator before weighing. The remaining solids were known as ash, and represented the fixed total solids. The weight lost represented the volatile solids. Total volatile solids offer a rough approximation of the amount of organic matter (plus plastics) present in the solid fraction of wastes. In particular, it was used in this work for determining if there was sufficient organic content for methanogenic activity.

Weighing for these tests was done as soon as possible, following cooling in a desiccator, as residues often are very hygroscopic and rapidly absorb moisture from the air. The total volatile solids in percentage of sample were calculated as:

$$\% \, Total \, volatile \, solids = (D\text{-}A)/(C\text{-}A)*100 \qquad (5)$$

A = weight of dish (g)
C = weight of dried residue + dish (g)
D = weight of residue + dish after ignition to 550°C (g).

4.2.5. Field Capacity

Since there are generally no accepted sampling and testing procedures for waste materials, the principles of general soil mechanics are usually applied despite the application of these principles to MSW being problematic as outlined in section 2.8.

Field capacity was measured using left over waste in the buckets after loading the test cells. Again, waste was mixed from multiple buckets to have a half bucket (nominally 8 kg) ready for compaction and moisture addition. The waste was weighed, then compacted following the same method for the test cells (refer Appendix C5 – Compaction). Water was then added until the bucket was full of water. It was left overnight to soak. The free water was then drained by gravity until drips fell at a rate of less than 6 per minute (one per 10 seconds). The sample was then reweighed and the wet field capacity calculated.

4.2.6. Particle Size Distribution

Particle sizing cannot be seen as representative of the waste in the landfill as the collection of waste sample from the landfill specifically excluded particles below 50 mm. Subsequent sub-sampling of this collected raw waste specifically exclude particles less than 20 mm. The

method adopted for sizing multiple raw waste samples was NZS 4402:1986 Test 2.8.2 Subsidiary method by dry sieving and one sample attempted following NZS 4402:1986 Test 2.8.1 Standard method by wet sieving.

4.2.7. Hydraulic Conductivity

Hydraulic conductivity was only measured through the test cells once the experiment had finished. Sivakumar (2013) showed there was reasonable agreement between measured total leachate volume/leachate flow and that calculated using Darcy's law. Darcy's law is valid for laminar flow through fine-grained sediments, as the dimensions of interstices are small and thus flow is laminar. Coarse-grained sediments also behave this way but the flow may become turbulent in very coarse-grained sediments, and Darcy's law is not always valid. In a fine grained soil like the sized, degraded MSW used in the experiment, geotechnical experience describes the falling head permeability test (AS1289.6.7.2-2001; ASTM D5856) as best. A schematic of this test is illustrated in Figure 17.

The cylindrical soil specimen has cross sectional area of A and length L. The standpipe has internal cross sectional area of a and overall length h. The falling head was measured between two points h_1 and h_2 on the standpipe over a measured time t. The hydraulic conductivity k can be calculated as:

$$k = \frac{aL}{At} ln \left(\frac{h_1}{h_2} \right)$$

Figure 17 Schematic of falling head permeability test setup

In geotechnical engineering k is commonly expressed in cm/s

The test cells subjected to the falling head permeability measurement had water added in excess until the water level remained above the top of the waste. This ensured saturation. The falling head test was then run on one low density and one high density test cell after the experiment was complete.

4.3.   Gas Parameters

4.3.1. Gas Analysis

Gas analysers are used capable of detecting methane (CH_4), carbon dioxide (CO_2), and oxygen (O_2), hydrogen sulphide (H_2S), carbon monoxide (CO) and hydrogen (H_2). Two different models of gas analyser were used (as one was found to be faulty during the experiment), the Geotechnical Instruments GA 5000 (shown as Figure 18) and the GA2000 Plus model. When the fault was suspected two sets of readings were taken in parallel to substantiate the error.

Both gas analysers have a water and particulate trap (Sartorius 0.22 μm filter) located on the inlet. The outlet runs off the gas analyser reconnected to the reactor to help regulate the headspace gas for analysis.

Methane and carbon dioxide readings were filtered to an infrared absorption frequency of 3.41 μm and 4.29 μm, respectively. Oxygen was measured by an internal electrochemical (galvanic) cell. Specification for these gases is given in Table 8 for the GA2000 Plus.

Figure 18 GA 5000 gas analyser

Table 8 GA2000 Plus specifications

Gas type	CH$_4$	CO$_2$	O$_2$
Range	0 – 100%	0 – 100%	0 – 25%
Gas accuracy	+ 3%	+ 3%	+ 1%
Response time	< 20 seconds	< 20 seconds	< 20 seconds

When the gas analyser was switched on, it performed a self-check and reported any requirements or malfunctions. Based on experience, the following steps were applied when using the gas analyser:

- The inlet filter prevents dirt, dust and water from entering and damaging the internal sensors and cells. This filter was always used and checked and replaced regularly.
- The instrument was left in the warmer environmental controlled laboratory containing the test cells for an hour before use to avoid condensation build-up in the tubing, filter or the instrument.
- Gas analysis was performed for a minimum of 60 seconds for each reactor, but held longer if necessary to ensure gas levels were stable.
- The equipment was purged with ambient air for two minutes before shutting down. The shut-down sequence also included an additional purge.
- The equipment was calibrated and certified regularly by laboratory staff.

4.3.2. Gas Volume

Gas volumes were determined by gas collection in one litre Tedlar bags (SKC Inc. USA) or via μFlow flow meter (manufactured by Bioprocess Control AB).

The Tedlar bags are equipped with a single polypropylene septum fitting which combines the hose/valve and the septum holder into one small fitting. Volumes from the Tedlar bags were measured by transferring the gas from the bag into a water displacement device (gasometer) as depicted in Figure 19. This gasometer consisted of an inverted 500 ml graduated cylinder sitting in a water column. It had an accuracy of ± 1% (5 ml). The bags were emptied of gas by pressing them gently by hand. The gas then flowed into the gasometer displacing the liquid (water), and in turn raising the inverted cylinder. The volume was read off directly by levelling the internal and external meniscus.
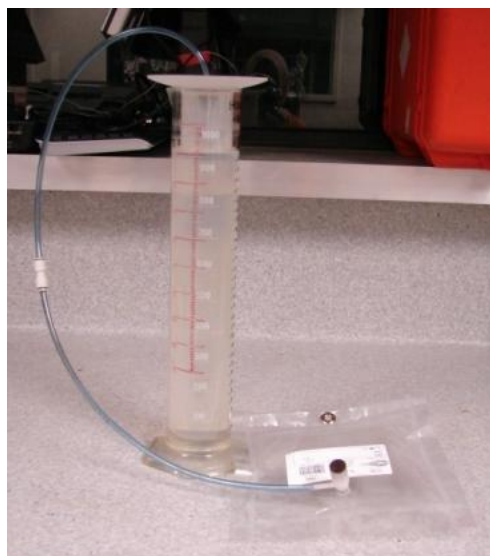
Figure 19 500 ml gasometer

The μFlow is a flow meter that works on the principle of liquid displacement and buoyancy and is designed for low gas flow detection. It featured automatic gas flow and volume normalisation. Measurement can range from 20 to 4000 ml/h with a resolution of 10 ml. One is shown in Figure 13 above the test cell on the plywood shelf.

Zaman (2010) used a similar process for measuring gas volumes and looked at the effect of dissolution of carbon dioxide in the tap water. She anticipated that the use of tap water would not be impacted by CO_2 loss because composition is measured directly from headspace prior to using the gasometer and pressure build up is avoided using Tedlar bags. Volume measurement from Tedlar bags via the gasometer takes seconds to complete.

4.4.   Leachate characteristics

4.4.1. Leachate Volumes

All leachate volumes (extracted from test cells and additional leachate added) were measured with graduated cylinders.

Uncertainty in measurement is half a division on either side of the smallest scale unit. However, the accuracy of measurements also depends on the quality of the glassware (e.g. Grade A or Grade B glassware). As Grade A measuring cylinders were not always available it was assumed uncertainty in leachate volumes was a whole division which is usually 1% for graduated measuring cylinders.

4.4.2. Solids Content

Total suspended solids (TSS) of leachate is often measured as part of operational control to reduce maintenance requirements caused by solids entrainment into moving parts, erosion, silt

build-up, and blockages. It is also a parameter measured in landfill experimental literature for determining settlement rate and degradation status. As solids content of leachate in this experiment was a function of the filtration capability of the waste and drainage layer, it was not of interest in this experiment.

4.4.3. Organics

Organics in leachate can be measured by via several methods depending on research aims; volatile solids, total organic content (TOC), chemical oxygen demand (COD), biochemical oxygen demand (BOD), and/or measurement of VFAs. Most research activities measure COD and VFAs. COD is readily determined and measures all the organic matter present (both biodegradable and non-biodegradable) and is considered more accurate than BODs. While reduced inorganic compounds can contribute to COD, the option to measure TOC was considered, and rejected, as it was less referenced than COD in literature. COD is also more recognised in landfill operation as highlighted by the ISWA Landfill Operational Guidelines (ISWA, 2019) which refer only to COD. Volatile solids are known to be less directly relevant to bioactivity and mathematical modelling but are an easier parameter to measure especially for verifying consistency and guiding experimental design and expectations.

VFA readings can be an insightful symptom that the reactor is edging towards a stressed or inhibited state. While monitoring VFAs is useful for reactor monitoring, it involves significant effort and consumables during sampling. Instrument degradation and malfunction is also common. This impacts the experimental timeline and experimental cost. (Zaman, 2010) indicates pH and methane percentage as the important indicators of reactor condition. A stable reactor is indicated by pH of 6.8-7.2 at 50% CH_4 and pH of 7.5-7.9 for 70% CH_4. A lower pH and methane percentage signified unhealthy reactors.

VFAs in the test cell leachate was analysed as a one off by Hills laboratory using Ion Chromatography and reported as sum of Formic, Acetic, Propionic and 1-8 Butyric acids only, expressed as acetic acid.

COD analysis of leachate was performed based on APHA standard methods section 5220 Chemical Oxygen Demand (COD) 5220 D. Closed Reflux, Colorimetric Method using the Hach Dichromate Reactor Digestion Method. The test was based on a standard curve developed in the laboratory (Hach Method 8000) with high regression coefficient at the range of testing (0.9993). Samples were digested in potassium dichromate and 50% sulphuric acid solution reagent. This was digested for two hours at 150°C, then determined colorimetrically at 620 nm wavelength on a Hach Spectrophotometer (program #435). Detectable COD concentration ranged from 0-1500 mg/L. Samples were diluted 1:1 with distilled water to ensure the soluble COD concentration was within the method calibration range. APPENDIX B – Protocols outlines the specific procedure for COD followed as set out by the CNRE laboratory.

COD for leachate was analysed on a filtered sample thus the value obtained was soluble COD and not total COD. The uncertainty of the COD test is given by the standard curve. Sensitivity of the measurements was 20 mg/L with 95% confidence level ±22 mg/L at 800 mg/L. Previous uncertainty measured by Zaman (2010) at 5% corroborated these uncertainty levels.

4.4.4. pH

pH close to neutral values is known to be best for anaerobic digestion systems. Zaman (2010) suggested pH was a good indicator of a distressed reactor, this has been verified by other researchers' findings.

pH was measured using an EDT RE357 Tx Microprocessor pH meter Series 3 and reported to one decimal place. The performance of the pH meter was continually checked using standard solutions at pH 4, 7 and 10 (Scharlau Chemie S.A., EU) and calibrated when required according to the instruction manual. When leachate volumes were too low (less than 10 ml) then pH-sensitive paper was used.

A sample of approximately 70 ml was collected in a 100 ml beaker from the graduated cylinder used to measure leachate volumes from the test cells. The sample was stirred 10 times with a glass rod, allowed to rest for 10 seconds and the probes inserted to a depth just off the bottom of the beaker. The pH was recorded when the reading became stable. Probes were rinsed with deionised water prior to taking the next reading. Samples were returned to the measuring cylinder they came from.

4.4.5. Hardness

Hardness was measured using EDTA Titrimetric method as outlined in APHA Standard Methods for Examination of Water and Wastewater 2340. Interference includes some metals which were checked against leachate metal analysis. The method states a relative standard deviation of 2.9% and a relative error of 0.8% based on sampling.

4.4.6. Sulphate (SO_4^{2-})

Sulphate levels in the leachate samples were measured using the Hach SulfaVer 4 method (Hach Method 8051) adapted from Standard Methods for Examination of Water and Wastewater. The procedure was equivalent to USEPA method 375.4 for wastewater. The method uses Barium chloride which reacts with sulphate ions present in the leachate to form a precipitate of barium sulphate. Using the Hach Spectrophotometer under program #680, the sulphate concentration between 2 – 70 mg/L can be detected at a wavelength of 450 nm. Dilution was not required for the leachate recovered from the test cells but was required for the raw leachate. Sensitivity of the measurements was 1 mg/L with 95% confidence level ±3 mg/L at 30 mg/L.

4.4.7. Phosphate (PO_4^{3-})

Reactive Phosphate levels in the leachate samples were measured using the Molybdovanadate Hach PhosVer 3 method (Hach Method 8048) adapted from Standard Methods for Examination of Water and Wastewater. The procedure was equivalent to USEPA method 365.2 for wastewater. The method uses ascorbic acid, which reduced the phosphate molybdate complex giving an intense blue colour. Using the Hach Spectrophotometer under program #490, the phosphate concentration between 0.02 – 2.50 mg/L can be detected at a wavelength of 880 nm. Dilution was required for some of the leachate recovered from the test cells and for the raw leachate. Sensitivity

of the measurements was 0.02 mg/L with 95% confidence level ±0.03 mg/L at 1 mg/L. Interfering substance levels indicated in the method include ferrous at > 100 mg/L.

4.4.8. Nitrate (NO_3^- -N)

Two nitrate readings were taken about midway through the experiment, two weeks apart. Reactive Nitrate levels in the leachate samples were measured using the Cadmium reduction Hach method (Hach Method 8039). The method reduced nitrate to nitrite using cadmium, producing an amber colour. Using the Hach Spectrophotometer under program #355, a nitrate concentration between 0.3 – 30 mg/L can be detected at a wavelength of 500 nm. Dilution was required for some of the leachate recovered from the test cells and for the raw leachate. Sensitivity of the measurements was 1 mg/L NO_3^---N with a 95% confidence level ±2 mg/L at 10 mg/L. Interference was experienced with chloride >100mg/L and at all levels of nitrite.

4.4.9. Nitrite (NO_2^-)

Two nitrite readings were taken about midway through the experiment, two weeks apart. Reactive Nitrite levels in the leachate samples were measured using the Ferrous Sulphate Hach method (Hach Method 8153). The method used ferrous sulphate to reduce nitrite to nitrous oxide to form a greenish-brown complex. Using the Hach Spectrophotometer under program #373, a nitrite concentration between 2 – 250 mg/L can be detected at a wavelength of 585 nm. Dilution was required for the raw leachate. Sensitivity of the measurements was 1.6 mg/L NO_2^--N with 95% confidence level ±7 mg/L at 200 mg/L

4.4.10. Ammoniacal Nitrogen (NH_3-N)

Two ammoniacal nitrogen readings were taken about midway through the experiment, two weeks apart. Reactive levels in the leachate samples were measured using the Salicylate Hach method (Hach Method 10031). Ammonia-compounds reacted with chlorine to form a monochloramine which reacted with salicylate to form 5-aminosalicylate which was oxidised in a sodium nitroprusside catalyst to form a blue compound, masked by yellow from the excess reagent to give a green-coloured solution. Using the Hach Spectrophotometer under program #343, the ammonia concentration between 0.4 – 50 mg/L can be detected at a wavelength of 655 nm. Dilution of 1:50 with distilled water was required for all samples of leachate recovered from the test cells and for the raw leachate.

Sensitivity of the measurements was 0.4 mg/L with 95% confidence level ±1.1 mg/L at 10 mg/L. Zaman (2010) used the same method with the same equipment and in repetitions found the uncertainty of the ammonia test to be 4%. Interference was experienced with >600 mg/L nitrate, > 5000 mg/L nitrite, and > 5000 mg/L phosphate.

4.4.11. Ferrous Iron(Fe^{2+})

Reactive ferrous levels in the leachate samples were measured using Hach 1,10 Phenanthroline method (Hach Method 8146) adapted from Standard Methods for Examination of Water and Wastewater. The method uses 1,10 phenanthroline indicator to react with the ferrous iron to form an orange colour. Using the Hach Spectrophotometer under program #255, a ferrous concentration

between 0.02 – 3.00 mg/L can be detected at a wavelength of 510 nm. Dilution was required for the raw leachate only. Sensitivity of the measurements was 0.010 mg/L with 95% confidence level ±0.011 mg/L at 1 mg/L. Interfering substances were not indicated although samples must be analysed as quickly as possible to avoid air oxidation of ferrous to ferric iron which was not detected.

4.5. Biomethanation Potential via Tube Reactor

The digested sewage sludge was sourced from a mesophilic digester at the Christchurch Wastewater Treatment Plant (WWTP). The WWTP digester input is a mix of primary and secondary activated sludge, from a typical city mix of residential, commercial, and small industrial sources. The 7 day old DSS is reported to have a pH of 7.2, 1% total solids (TS) with 69% being volatile solids. (VS)

Zaman (2010) established substrate loading quantities by reactor volume empirically, based on methane yield. A maximum organic loading of 25.1 g volatile solids per litre per day (g VS/L) is tolerable for food waste, but an organic loading of 18.8 g VS/L was suggested to be more stable, as higher methane yield can be achieved. She also noted that an organic loading of up to 240 g wet weight food waste was viable. White rice is the preferred organic matter and performed consistently in the experiments.

The quantity of solid waste material used here needed to align with the 18.8 mg VS/L recommended and was quantified using volatile solids methods as described in section 4.2. Buffiere, Loisel et al. (2006) note that describing organic matter degradation using volatile solids alone is limited. Type, as well as quantity, of degraded volatile solids affect methane production. Fats, carbohydrates, or proteins have different methane potential, as does the chemical nature of the specific volatile solid type; for example cellulose is known to be more difficult than simple sugars to breakdown.

Based on a target volatile solids level of 18.8 grams volatile solids per litre (18.8 gVS/L), the required quantity of raw waste was calculated to be 855 g. This is based on an average volatile solids content on the raw waste of 8.8%, a moisture content of 25% and using 1 litre of DSS per tube reactor.

The test result can be obtained within 4 days but improved data is gained at around 20 days. These represent averages of 50% and 95% methane production respectively. Each test run herein was operated for 34 days, at which point there ceased to be any significant change in the observed gas production. this aligns with many standard tests for methane production.

Zaman (2010) outlines in detail the process for the experiment using the tube reactors. Setup consists of the following steps:

1. Weigh waste
2. Load waste into tube
3. Measure seed (1 litre DSS)
4. Load seed into tube
5. Stir mixture

6. Close lid tightly and all valves
7. Flush headspace with nitrogen gas
8. Read gas composition using gas analyser
9. Weigh tube
10. Connect Tedlar bag or gas meter and incubate at 35-40ºC

The tube reactors were stored in a temperature controlled room heated to a mesophilic temperature of 35°C ± 3°C for the duration of each test. The gas composition of the headspace of each tube reactor was determined with a landfill gas analyser (Geotech GA 2000 Landfill Gas Analyser). The percentage of methane, carbon dioxide, and oxygen were recorded daily by connecting the gas analyser to both the inlet and outlet for each tube reactor and operating the pump for approximately 50-60 seconds.

Daily data collection consisted of measuring the gas composition and volume as follows:

1. Shake tube on a 45º angle for 40 seconds
2. Measure gas composition using gas analyser*
3. Measure gas volume by pressing biogas collected in Tedlar Bags into gasometer.

The experiment was designed to determine if the waste sampled from the test cells contained organic matter that could be readily converted to methane. This would assist in determining a possible explanation for the failure of the experiment to produce gas.

To ensure consistency and effectiveness in experimental setup and operation, and to ensure results would confidently determine organic potential for methane production as expected, a series of tube reactor tests were run as shown in Table 9.

<p style="text-align:center">Table 9 Tube reactor experiments undertaken</p>

Tube reactor experiment	Contents	Reason for experiment
Blank (x4)	DSS only. No substrate	QA no substrate Background level of methane for DSS
Rice (x4)	Rice	Background level for rice QA substrate effective
Raw waste (x5)*	Raw waste sampled from Kate Valley	Baseline of organic content of waste sampled
Spent waste (x6)	Spent waste from test cell 3 or test cell 5 post experiment	Determine extent of organic content loss from raw waste
Refresh waste (x3)	Refreshed waste from Kate Valley	Determine extent of deterioration due to waste storage since start of experiment
Raw waste + rice (x2)	Raw waste + rice	Verification that organic content is the cause (not inhibitors)
Refresh waste+ rice (x2)	Refresh waste+ rice	Verification that organic content is the cause (not inhibitors)

* Tests were run in 2 batches as there was a finite number of tube reactors available. Results are treated together.

5. EXPERIMENT RESULTS

5.1. Waste Parameters

5.1.1. Test Cell Parameters

Figure 20 shows a photo of the actual experimental setup in its support frame in the environmental laboratory.

Figure 20 Experiment showing four test units (eight test cells) in their frame

The following table (Table 10) is a summary of test cell parameters once the waste was loaded.

Table 10 Test cell waste parameters

Waste Parameter	Test cell							
	1	2	3	4	5	6	7	8
Type	Low density intercept	High density intercept	Low density std	High density std	Low density intercept	High density intercept	Low density std	High density std
Weight (kg)	42.1	59.8	41.3	64.5	43.3	57.5	46.9	64.6
Density (kg/m^3)	1007	1451	1003	1565	1066	1396	1121	1568

The average density of the low density test cells was 1049 kg/m^3. The average density of the high density test cells was 1495 kg/m^3.

5.1.2. Waste Composition

The average composition of the waste sampled (> 10 mm), by category is illustrated in Table 11 and Figure 21. Raw results of the seven samples taken and sorted appear in Appendix D2 – Waste Characteristics.

Table 11 Waste composition

Type	%
Clay or soil	48.9%
Stones	25.8%
Glass & ceramic	5.4%
Paper and cardboard	4.6%
Wood	4.6%
Plastic (except foam)	3.7%
Other	2.6%
Textiles	1.6%
Metal	1.6%
Ravelled fibres	0.8%
Foam	0.5%
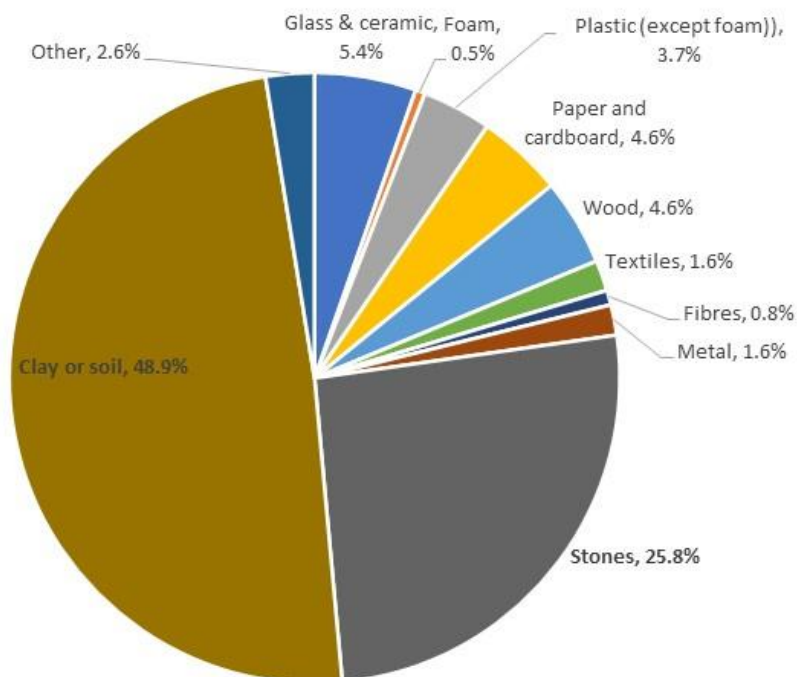
Figure 21 Waste composition

There appeared to be a larger proportion of clay and stones than expected. This can be partially explained by considering that this is a weight based analysis and the sample sizing is less than 50 mm and greater than 10 mm. This has the effect of excluding large light plastic bags and small biodegradables in the sample, yet these types of waste were apparent in the landfill (see Figure 11).

5.1.3. Moisture Content

Waste samples were fairly dry to touch, and their colour was dark brown. After drying, the sample colour appeared medium to light brown. In texture, the waste resembled a highly organic sand with fair amounts of granular and fibrous materials.

Moisture content of the raw waste was measured to be 23%. Samples were remarkably consistent and Figure 22 presents the results as a Pareto chart to show this consistency in results. Moisture content varied between 19% and 25% over 24 samples with two additional outliers of 17% and 35%.
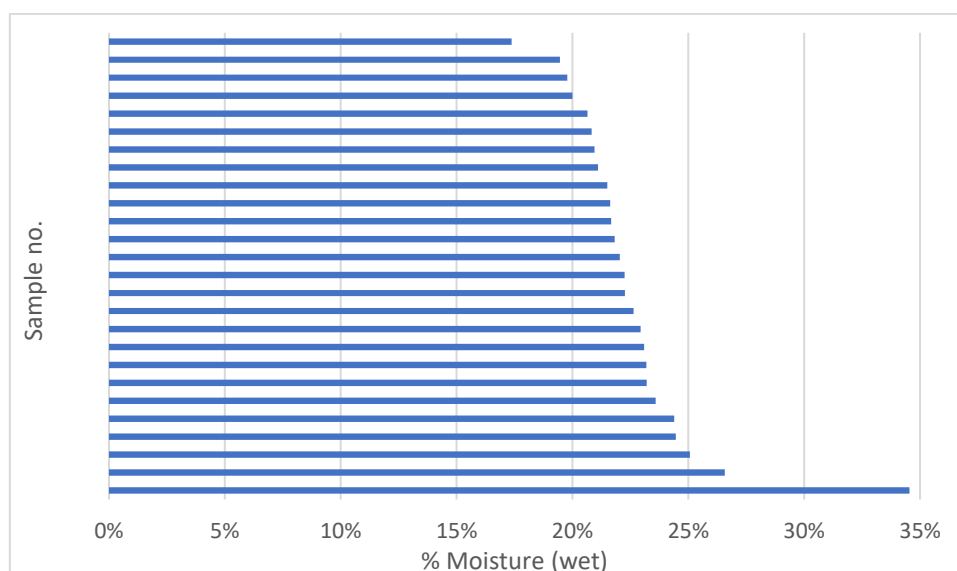
Figure 22 Moisture content of waste (Pareto)

5.1.4. Field Capacity

Field capacity was initially measured using three samples of low density and three samples of high density waste based on the compaction process as described in Appendix C5 – Compaction. Average field capacity estimates were 38% for the low density waste samples and 31% for the high density waste samples. Samples varied up to 9% as shown in Figure 23 over the three samples for both densities.
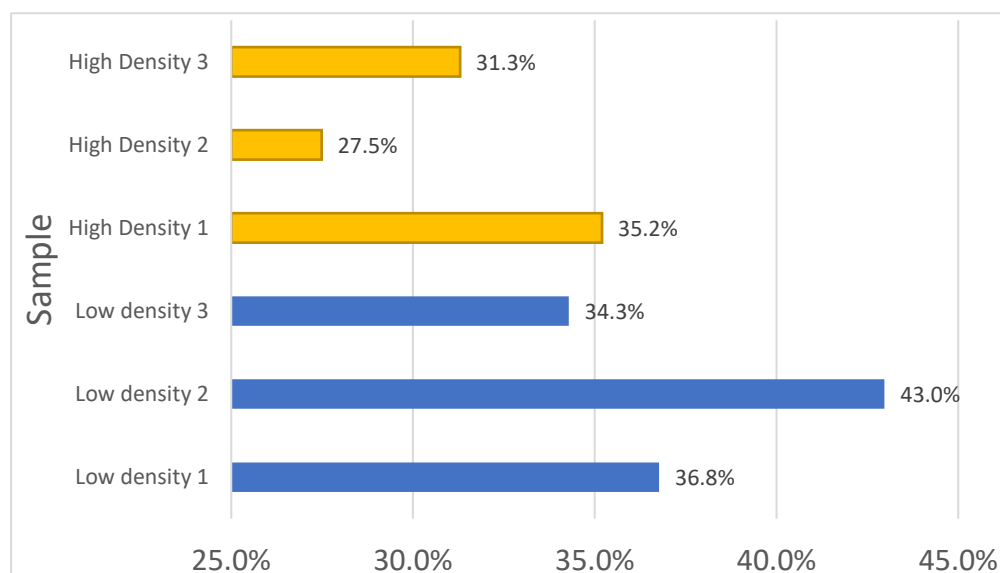
Figure 23 Initial field capacities based on compacted waste samples

Initial sampling to obtain field capacity figures was necessary to determine the theoretical volume of leachate that would need to be added to the test cells to ensure field capacity was met and biodegradation optimised. Table 12 summarises these values.

Table 12 Leachate addition requirements based on initial estimated field capacity

Field capacity parameter	Test cell							
	1	**2**	**3**	**4**	**5**	**6**	**7**	**8**
Estimated density (kg/m^3)	1020	1632	1020	1632	1020	1632	1020	1632
Type	Low density intercept	High density intercept	Low density std	High density std	Low density intercept	High density intercept	Low density std	High density std
Actual density (kg/m^3)	1007	1451	1003	1565	1066	1396	1121	1568
Initial estimated Field capacity	38%	31%	38%	31%	38%	31%	38%	31%
Estimated leachate requirement (l)	6.32	4.78	6.20	5.16	6.50	4.60	7.03	5.17

5.1.5. Hydraulic Conductivity

Hydraulic conductivity was measured multiple times on one low density and one high density cell on completion of the experiment. A summary of the outcomes are presented in Table 13.

Table 13 Hydraulic conductivity results summary

	Test cell 7	**Test cell 6**
Density	low density	high density
Sample count	24	24
Average hydraulic conductivity	1.60×10^{-4} m/s	2.85×10^{-5} m/s
Range	$1.57 - 1.62 \times 10^{-4}$ m/s	$2.54 - 3.29 \times 10^{-5}$ m/s
Standard deviation	1.71×10^{-6} m/s	2.20×10^{-6} m/s

5.1.6. Total Volatile Solids

The raw waste collected from Kate Valley had an average total volatile solids content of 9.2% by dry weight over the multiple subsamples measured from three of the collected buckets.
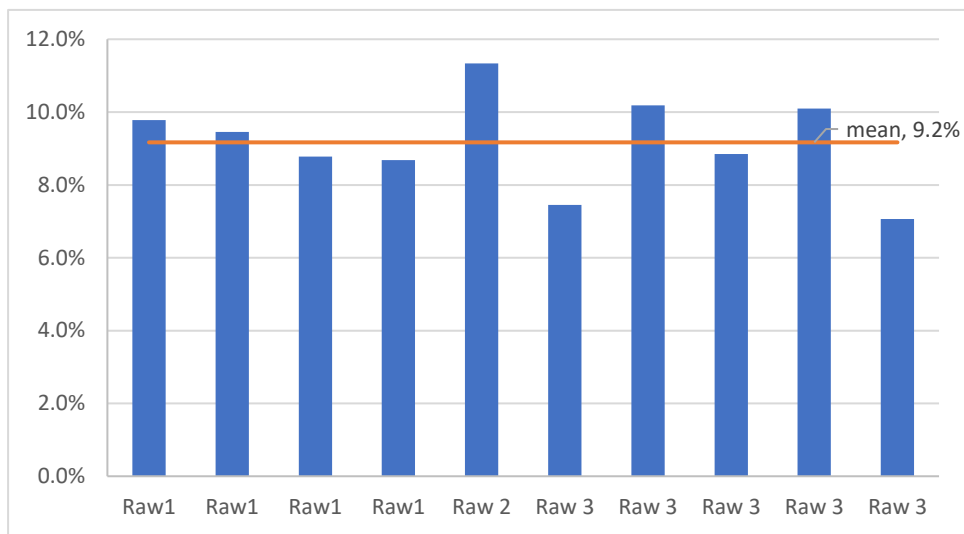
Figure 24 Volatile solids ranges of three samples of raw waste

The volatile solids measured from samples taken from each of the test cells at the experiment's conclusion averaged 8.8%, with approximately the same variation as the original samples.
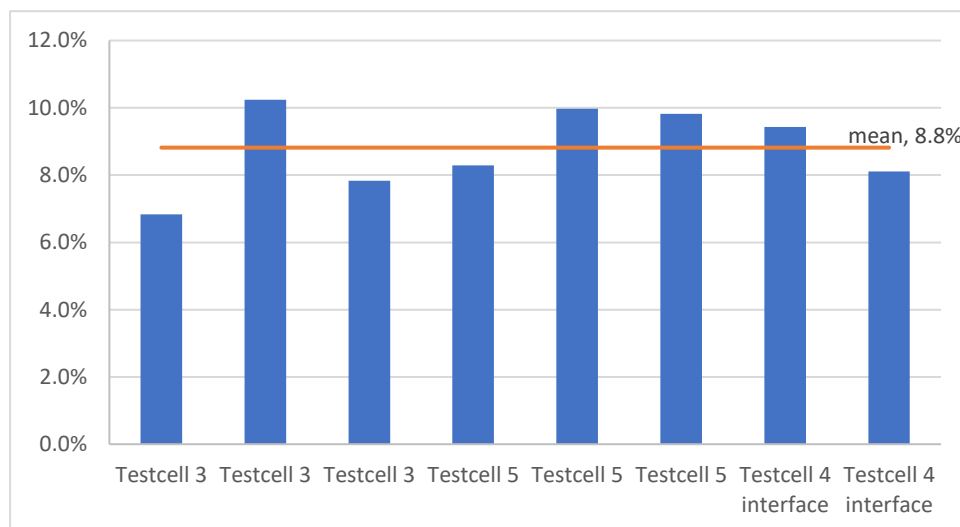
Figure 25 Volatile solids of three test cells at the end of the experiment

5.1.7. Waste Organic Content

Four waste samples (<2mm) were sent to Hills laboratory for determination of COD. Table 14 presents the results.

Table 14 COD of Waste

Sample id	Chemical Oxygen Demand (COD) (mg O_2/kg dry waste)
Sample 1	8,500
Sample 2	16,200
Sample 3	35,000
Sample 4	23,000
Mean	**20,675**

The determination of the biomethane potential used 26 tube reactor experiments in two batches. Appendix D6 – Biomethanation Potential Results presents the methane levels recorded and gas volumes measured. Table 15 summarises the results showing ranges and averages.

Table 15 Biomethanation potential summary results

Tube contents	Average volume of gas produced (ml)	Range of volume (ml)	Maximum methane level achieved (%)
Blank	175	70 - 285	11.3
Rice	3415	1165 - 4697	65.4
Raw waste	1037	31 - 2027	45.4
Spent waste	210	0 - 320	13.5
Refresh Kate Valley	5710	3222 - 8198	73.0
Raw waste + rice	1898	1510 - 2285	71.4
Refresh waste+ rice	9167	4295 - 14038	78.9

5.1.8. *Particle Size Distribution*

NZS 4402:1986 Test 2.8 Determination of Particle Size Distribution is specifically for soil and states the dry sieving method is not to be used unless it has been shown that for the type of material being tested, the same results are achieved using the method for wet sieving. Wet sieving was attempted but this method causes excessive ravelling and particle breakdown of the sample (see Figure 26), blocking the sieves less than 1.18 mm.

Figure 26 Example of fluff in sieved sample

So, despite the of requirements of NZS 4402, only dry sieving was used. Five samples were completely dry sieved from 50 mm to 75 µm. Figure 27 shows the combined result of these size distributions and the fairly tight range of results. Figure 27 also shows the waste sampled can be considered to have similar particle sizing to sand.

CLAY	SILT			SAND			GRAVEL	
	Fine	Medium	Coarse	Fine	Medium	Coarse	Fine	Medium
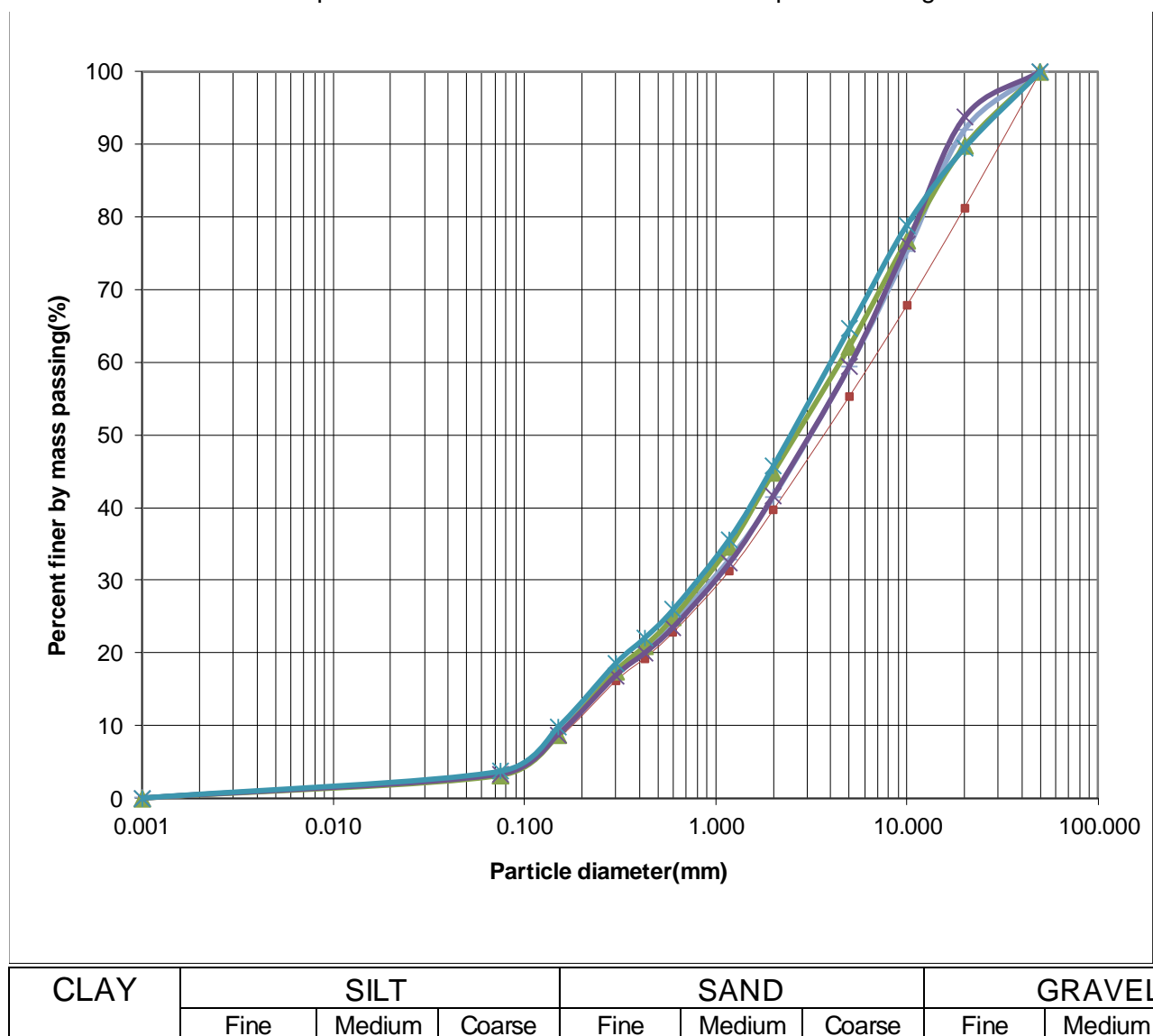
Figure 27 Particle size distribution curve (50 mm-75 µm)

Figure 28 zooms into the fine portion (75 μm to 1.18 mm) of this sizing distribution. Subsamples of the samples used for the sizing results above could be created for sizing less than 2 mm. This is because the standard (NZS 4402) gives a maximum mass of material to be retained on each test sieve and this is substantially less at lower sieve sizes. This fine portion is based on a total of 12 samples.
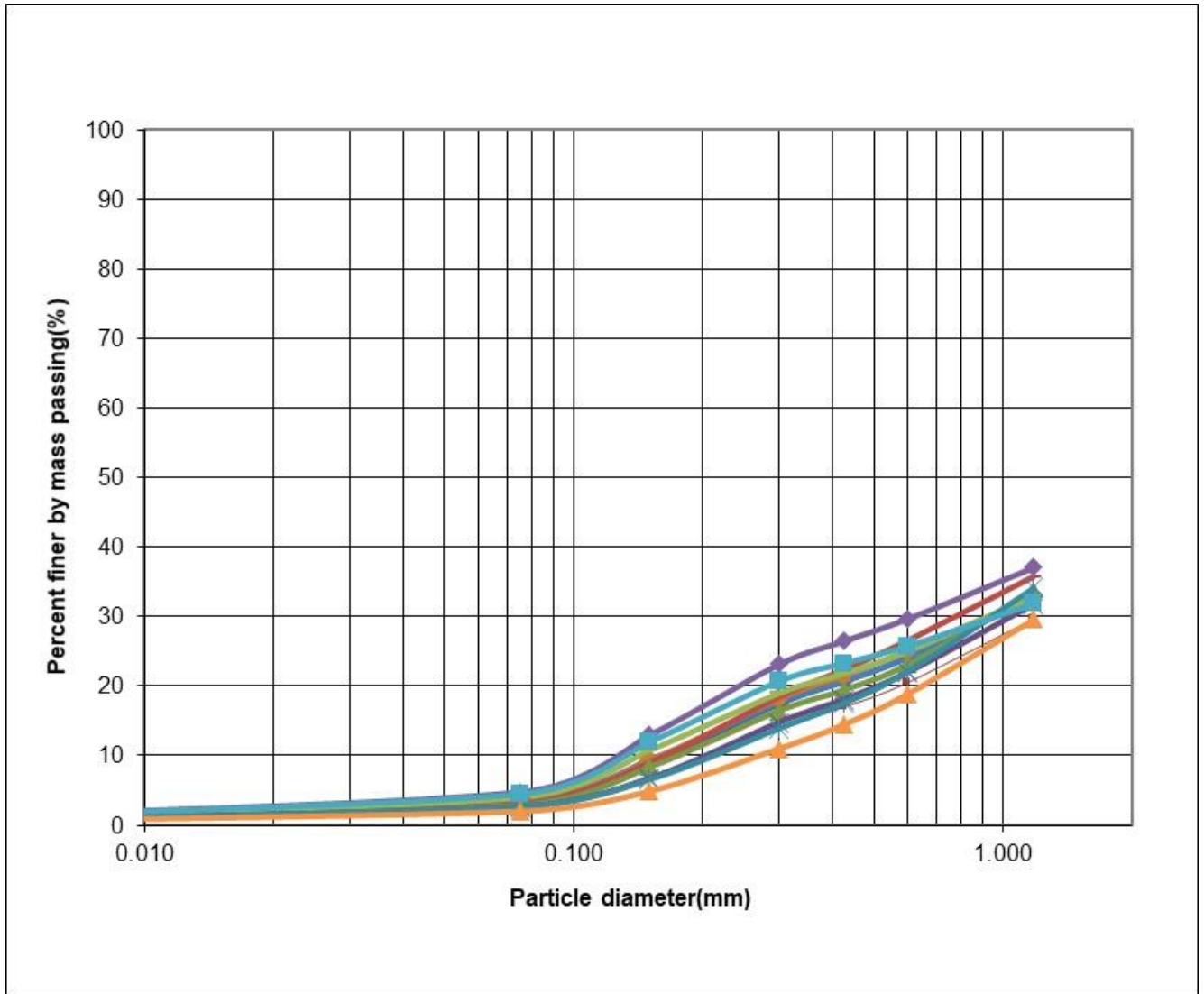
Figure 28 Particle size distribution curve (1.18 mm-75 μm) – expanded abscissa

The final particle sizing distribution measurement undertaken was to take a dry sieved sample at less than 2 mm and re-sieve it to qualify the fragility of the particles. Figure 29 shows the results of these two size distributions (sieved and immediately re-sieved).
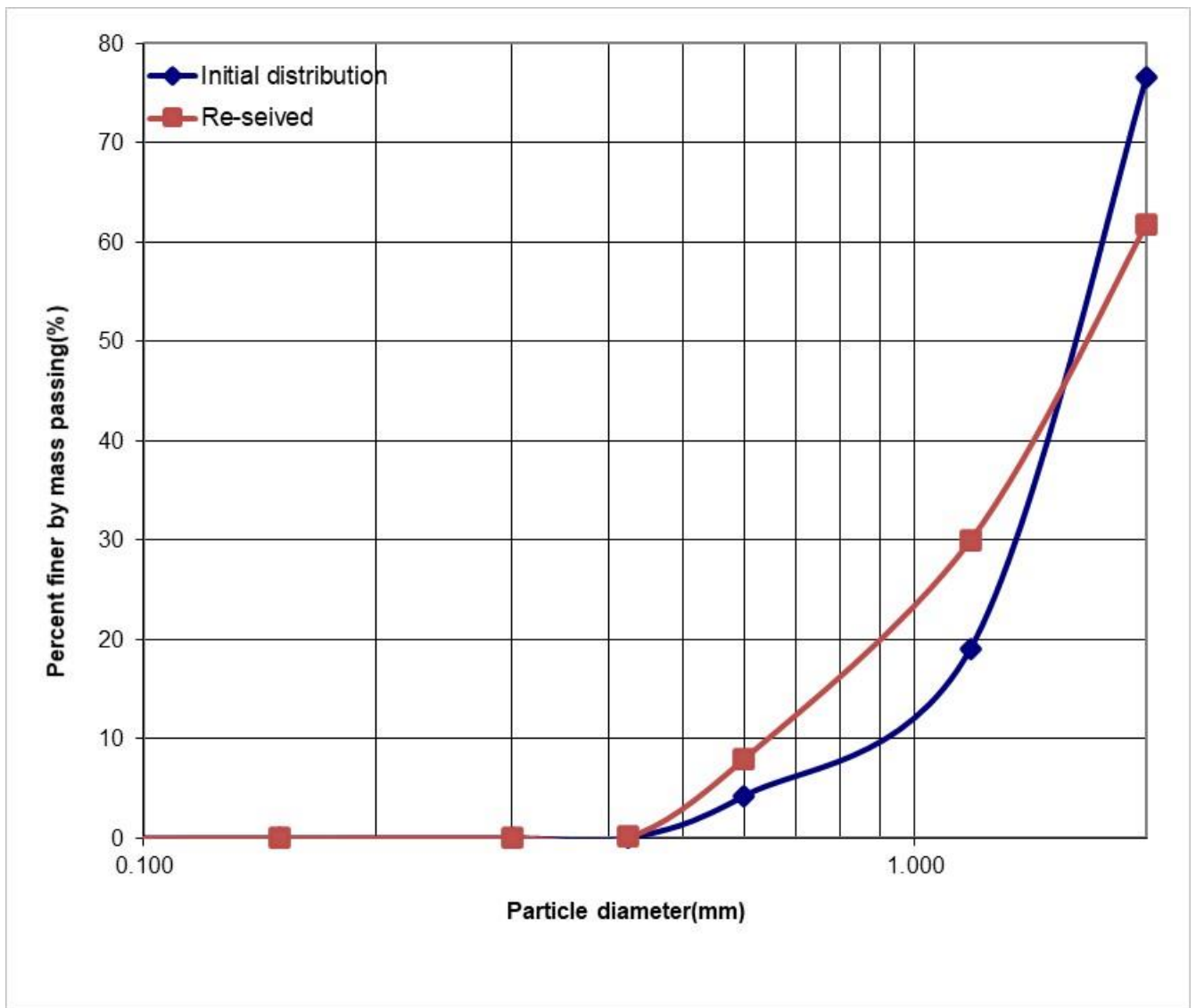
Figure 29 Particle size distribution showing particle breakdown during re-sieving (expanded)

It is apparent there is up to a 19% breakdown in particle size. This appears immediately on the first sieve (1.18 mm) and to a lesser extent on subsequent sieves. This result reinforces the breakdown of particles apparent during attempts at wet sieving.

5.1.9. Settlement Post Experiment

When emptying the test cells it was noted that they all exhibited a decrease in waste height visible from both the outside and inside. This may be due to some settlement perhaps due to the biocompression or mechanical creep as referred to by Gao, Bian et al. (2017). It was difficult to measure or define, as while a "tide mark" existed, some pooling especially on the high density test cells had occurred. The difference on height was not statistically significant (i.e. within an error of measurement of ±5 mm) but appeared consistently lower in all test cells. The maximum consolidation observed was 5mm/350mm = 1.4%.

5.2. Leachate Results

5.2.1. Leachate Volumes

Once the test cells were loaded and purged with nitrogen to exclude oxygen, six litres of leachate were added over the first five days to all test cells. On day six, test cells 1, 2, 3, 5, 6, and 7 all had leachate exiting the test cell. Test cell 6 still had a pond of leachate on top of the waste. Test cell 4 had leachate in the drainage layer on day eleven but also retained a pond on top. Test cell 8 had leachate ponding throughout the experiment and leachate did not reach the drainage layer until day 60.

The experiment to allow the waste to degrade ran for 163 days with leachate added and recirculated and gas composition measured daily for the first week then every second or third day until the end of the experiment. Leachate quality tests were also performed intermittently and gas volumes were monitored despite no gas being produced. Table 16 summarises leachate flow results.

Table 16 Test cell leachate flow results

Leachate volume Parameter	Test cell							
	1	2	3	4*	5	6	7	8*
Type	Low density intercept	High density intercept	Low density std	High density std	Low density intercept	High density intercept	Low density std	High density std
Leachate breakthrough (days)	6	6	6	11*	6	6*	6	60*
Average leachate flow (ml/day)	74	26	110	27	110	43	85	14

* ponding apparent on top of the waste.

The quantities of leachate added, recirculated and withdrawn from the test cells was measured and this enabled the volume of leachate held up, or retained in the test cells to be tracked over the period of the experiment. Figure 30 and Figure 31 summarise leachate retained volume data from all test cells split by waste density.
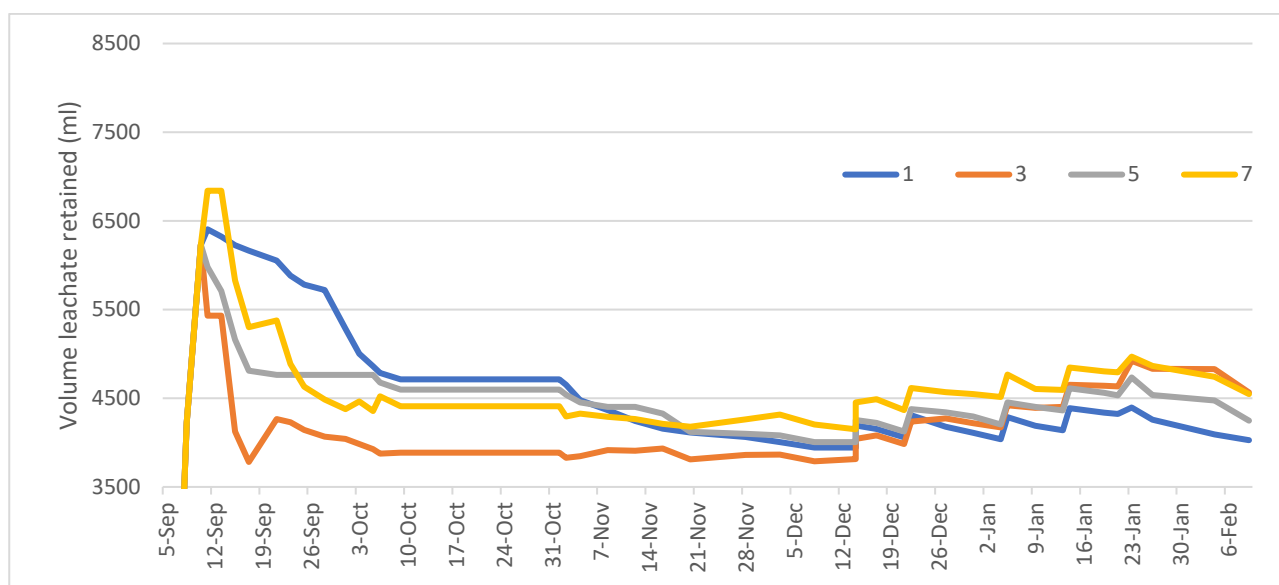
Figure 30 Retained leachate volumes – Low density test cells
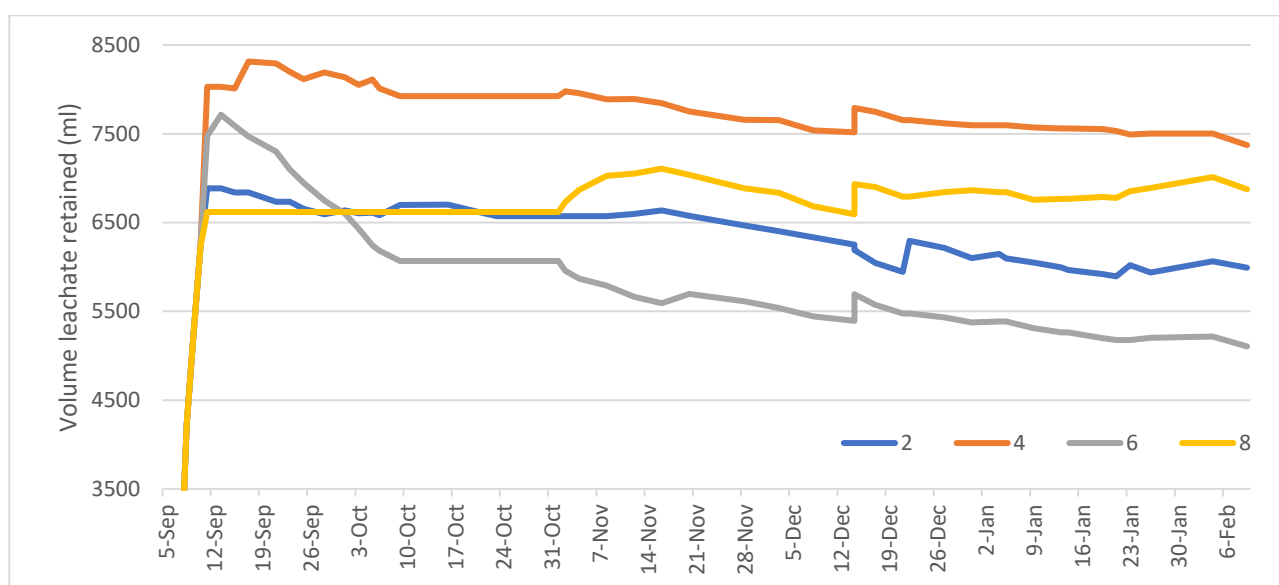
Figure 31 Retained leachate volumes – High density test cells

5.2.2. pH

The raw leachate from the landfill used to get the waste up to field capacity and top up was at pH 7.8-7.9

pH was measured from day six, (15 Sept) providing the cells had produced leachate in sufficient quantities to measure using the pH meter (greater than 40 ml). Test cell 4 did not produce leachate until day eight (17 Sept) and test cell 8, day 60 (4 Nov).

None of the test cells were producing gas by day 40 so all test cells were left undisturbed for 15 days. No leachate was withdrawn and no pH was measured. Figure 32 charts the pH over the duration of the experiment and shows the period when no measurements were made.
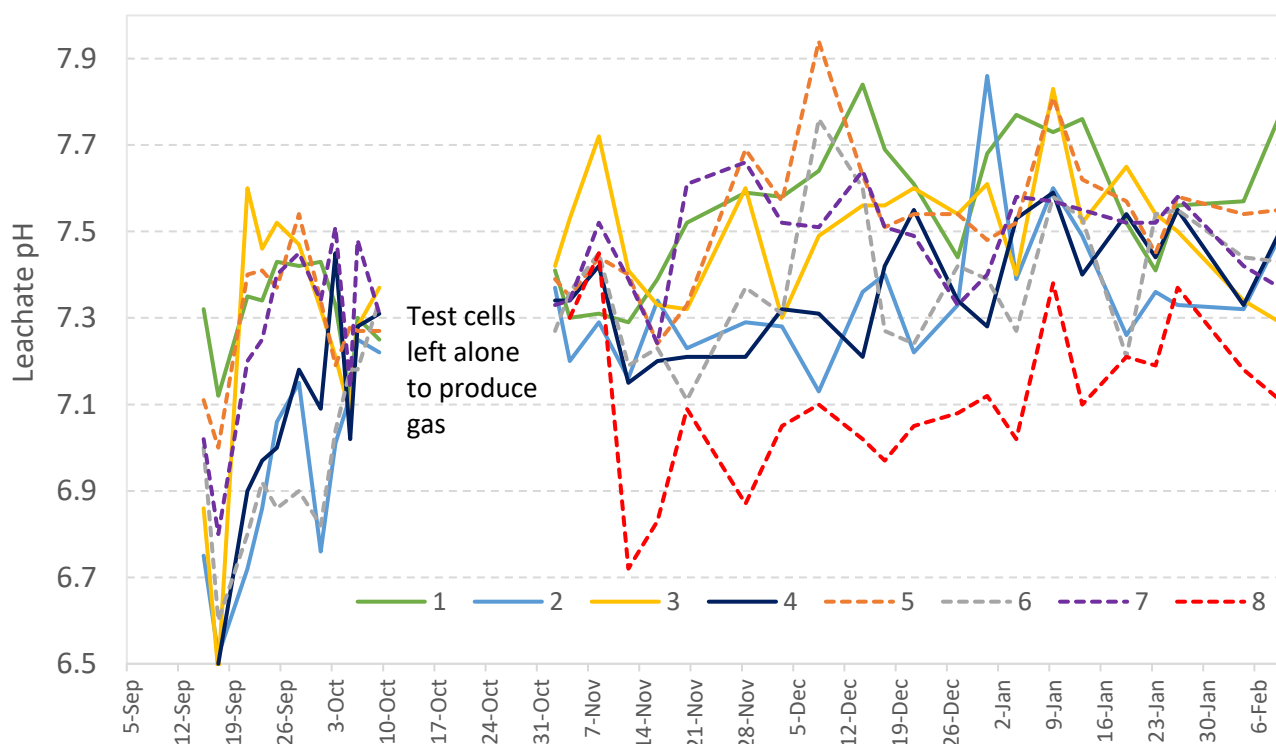
Figure 32 Leachate pH for all test cells

All test cell leachate sits within pH 7.1 to 7.9 once established except for test cell 8, which had ponding for most of the experiment.

5.2.3. Oxygen Demand

Raw leachate COD levels averaged 3540 mg/L varying between 3300 to 3800 mg/L across samples and throughout the laboratory study. Hills laboratory verification analysis reported similar levels.

Figure 33 outlines average COD levels for each test cell. Figure 34 shows the data over time.
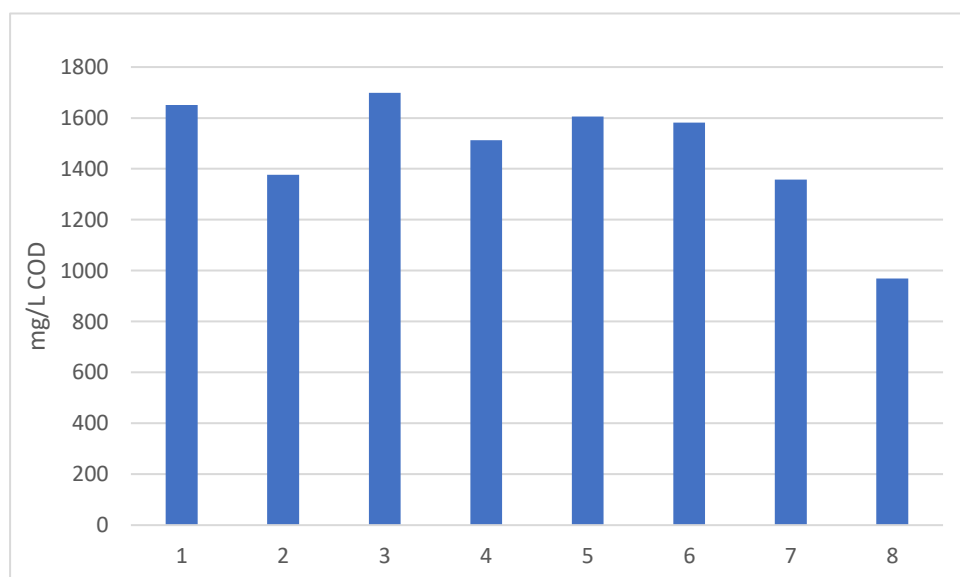
Figure 33 Leachate COD levels (averaged over the experiment)
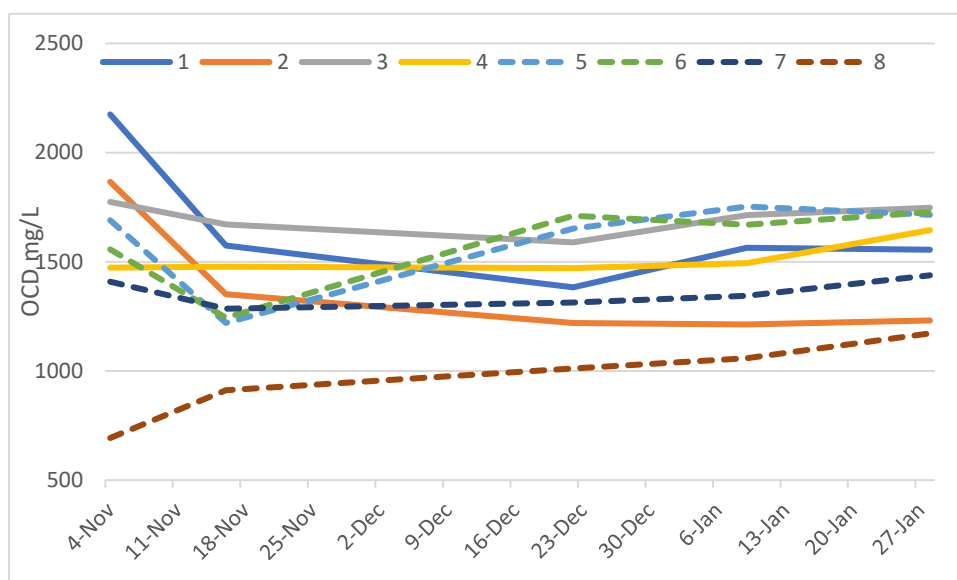
Figure 34 COD levels of test cells over time

5.2.4. Nitrogen Analysis

Nitrate, nitrite, and ammonia-nitrogen levels were measured for all test cell leachates. Except for test cell 7 all other test cells had slightly higher readings for nitrite and ammonia between the two samples (two weeks apart). Total Kjeldahl Nitrogen (TKN) levels were analysed for all test cells by Hills Laboratory.

Table 17 Nitrogen levels in test cell leachate (Hach method)

Test Cell	NO_3^- (mg/L)		NO_2^- (mg/L)		NH_3 - N (mg/L)		TKN (Nmg/L)
1	12	24.3	65	146	505	560	570
2	18.7	3	4	25	260	270	270
3	21.3	29.7	75	137	630	710	510
4	4.8	13.2	80	158	415	470	400
5	14	14.2	69	89	695	745	680
6	17.9	8.6	67	94	460	465	450
7	4.4	3.6	81	74	445	435	440
8	1	1	26	22	225	260	155

It should be noted that nitrate analysis is interfered with by chloride (at levels greater than 100 mg/L) and by nitrite (at all levels). Both these conditions were experienced, meaning the nitrate levels must be treated with wariness. Raw leachate levels tested by this method were found not to be reliable as readings fluctuated greatly between duplicate samples and required variable dilution across different samples over the two weeks. This is thought to be due to interfering analytes and time spent in storage since collection.

It is not anticipated that there were any interfering chemicals with ammoniacal nitrogen or nitrite analysis.

5.2.5. Other Analytes

Sulphate was measured for all test cell leachate and was non-detectable, indicating levels less than 2 mg/L. Sulphate in the raw leachate was also less than 2 mg/L. All other analytes were within detection ranges.

Table 18 Levels of other analytes in raw and test cell leachates (Hach methods)

Test Cell	PO_4^{3-} (mg/L)	Fe^{2+} (mg/L)	Total Fe (mg/L)	SO_4^{2-} (mg/L)	Cl^- (mg/L)
1	2.10	0.61	1.88	<2	1106
2	1.48	0.87	3.16	<2	1205
3	2.38	0.75	2.63	<2	1146
4	1.42	0.02	2.53	<2	1164
5	2.64	0.59	2.31	<2	1194
6	1.07	0.61	3.12	<2	1309
7	1.85	0.56	2.6	<2	1093
8	1.48	0.21	3.89	<2	1110
Leachate	32	0.1	*	<2	1976

A sample of raw leachate was sent to Hills laboratory for verification analysis. The results are presented in Table 19.

Table 19 Raw leachate chemical analysis (Hills Laboratory)

Analyte	Level	unit
Total Suspended Solids	31	mg/L
Total Dissolved Solids (TDS)	3700	mg/L
Settleable Solids	< 1.0	mL/L
Total Calcium	46	mg/L
Dissolved Iron	0.9	mg/L
Total Iron	1.07	mg/L
Total Magnesium	30	mg/L
Total Potassium	350	mg/L
Total Sodium	800	mg/L
Chloride	970	mg/L
Total Ammoniacal-N	920	mg/L
Nitrite-N	< 0.10	mg/L
Nitrate-N	< 0.10	mg/L
Nitrate-N + Nitrite-N	< 0.10	mg/L
Dissolved Reactive Phosphorus	8.1	mg/L
Phosphate	25	mg/L
Sulphate	7	mg/L
Total Biochemical Oxygen Demand (BOD)	25	mg O_2/L
Chemical Oxygen Demand (COD)	1720	mg O_2/L
Dissolved Organic Carbon (DOC)	640	mg/L
Total Organic Carbon (TOC)	670	mg/L
Volatile Fatty Acids (VFA)	< 10	mg/L
Formic Acid	< 5	mg/L
Acetic Acid	< 5	mg/L
Propionic Acid	< 5	mg/L
Butyric Acid	< 5	mg/L

5.3. Gas Composition

Anaerobic biodegradation can be evidenced by gas composition. Gas flow is known to start when methane levels get above 50-60% and flow is maintained if the level remains above 40-50%. Methane and carbon dioxide levels measured by test cell appear in Appendix D3 – Gas Results. Figure 35 summarises this for duplicate test cells.
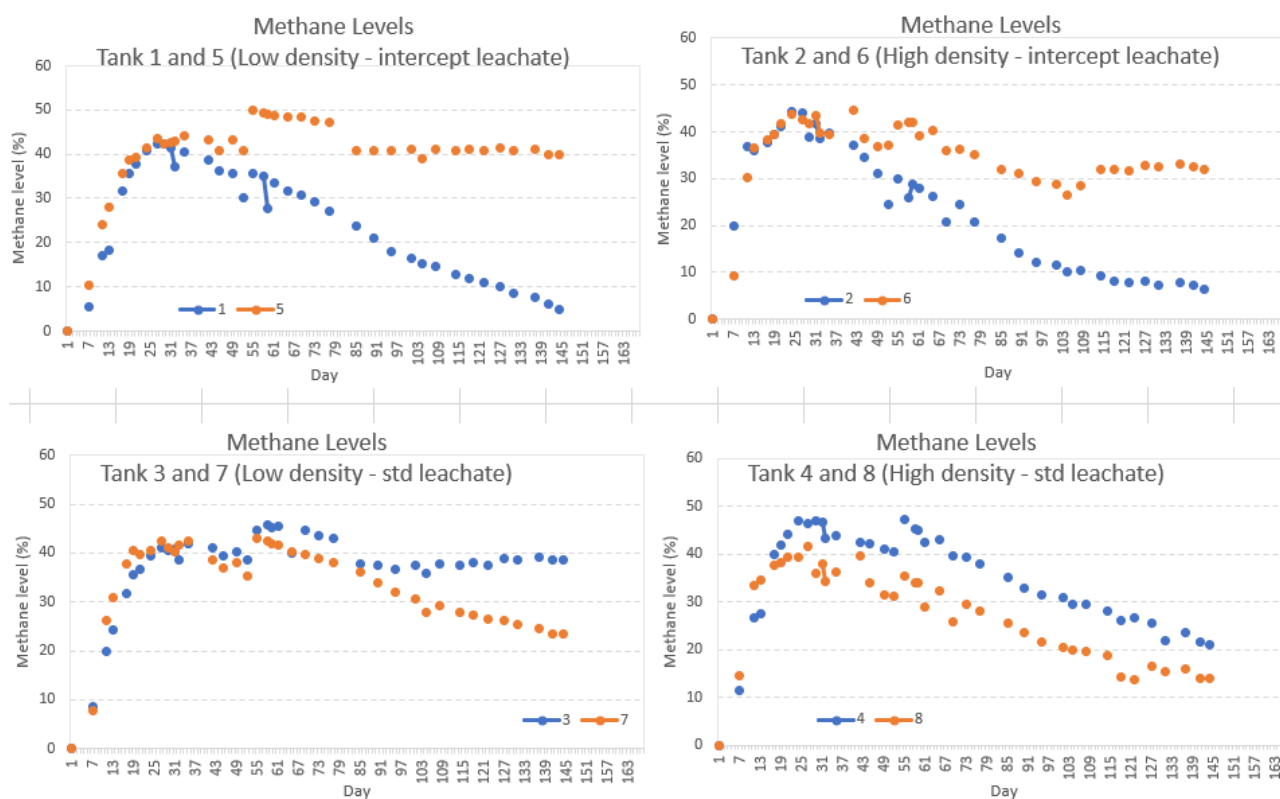
Figure 35 Methane and carbon dioxide levels for all test cells by duplicates

This shows no test cells ever obtained over 50% methane.

No gas flows were apparent for any of the test cells over the 163 days of the experiment.

Oxygen levels and hydrogen sulphide levels were also recorded. Oxygen levels data by test cell also appear in Appendix D3 – Gas Results, but Figure 36 shows all the oxygen levels recorded for all test cells and clearly shows the result of swapping out the gas analyser on October 29. Oxygen levels dropped when the new calibrated gas analyser was used.
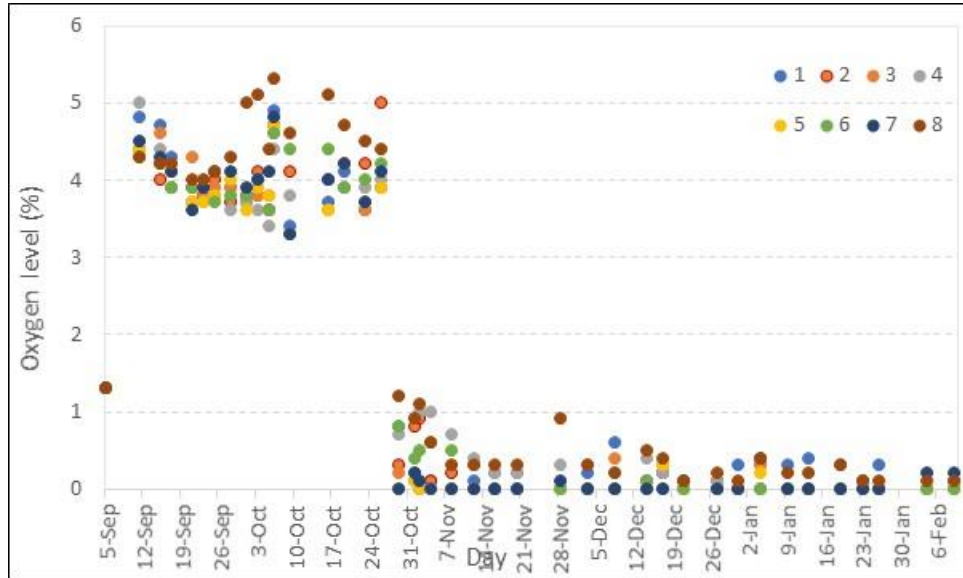
Figure 36 Combined oxygen levels in gas analysis showing equipment error

Both gas analysers were used to measure gas levels for the two periods of the swap. Appendix D3 – Gas Results compares this data and section 6.2.4 discusses this error in more detail.

Hydrogen sulphide results are presented in Figure 38 and discussed with respect to sulphate inhibition potential in section 6.2.2.

6. DISCUSSION

6.1. Conditions for Methanogenesis

Methane indicates the presence of methanogenic bacteria. Gas compositions charted in Figure 35 shows methane levels were significant but never reached 50% for any test cell. Typically, test cells do not produce enough gas until methane levels approach 60%. Methane levels experienced can also be seen to fall in Figure 35 after approaching 50% in all test cells after approximately 30 days. No gas was released from any test cells. This is indicative of either problematic experimental conditions, chemicals which inhibit methanogenic activity, or insufficient organic material. The fact that all test cells behaved the same way indicates that the cause of inadequate gas production is likely to be common.

Biodegradation of organic material is evident by the presence of methane and carbon dioxide. Anaerobic biological activity is primarily evident by gas composition, volatile acid production and pH changes. Figure 32 in section 5.2 indicates pH levels were favourable for methanogenesis (pH=7 to 8) for all test cells (except perhaps test cell 8 in early stages). It can be seen from Figure 32 that test cell 8 has a pH consistently below the range of the other test cells. Figure 37 shows the pH of test cell 8 and its duplicate test cell 4. Both these test cells had ponding. For test cell 8 it lasted the bulk of the experiment. Its trend to increasing pH is an indication that the cell is not unhealthy or souring.

Figure 37 Standard recirculation high density test cells (4 and 8) pH levels

Strong reducing environments are indicated by:

- the absence of nitrate
- the reduction of sulphate
- high values of soluble reduced iron

In anaerobic conditions nitrification is inhibited (Lu, Eichenberger et al., 1984) and the NH_4-N form predominates. This is experienced in the test cells (see Table 17) and is equally apparent in the raw leachate (Table 19). Nitrate appears to be present in the test cell leachates at low levels (Table 17), and these levels are higher than the raw leachate (Table 19) indicating the waste is

contributing to raise the nitrate levels. Although it is known the testing procedure experienced interference due to levels of nitrite and chloride, making this inference not definitive.

Sulphate levels are not detectable in the test cell leachates (Table 18) and appear in only low levels in the raw leachate (Table 19). Soluble iron was not experienced (Table 18, Table 19).

6.2.   Inhibitors

6.2.1. Overall Indicators

The test cells all reported significant methane and companion carbon dioxide levels in the headspace, indicating inhibition of methanogenesis is unlikely. However it is recognised that some inhibition may be causing these levels to stagnate and this was investigated.

6.2.2. Sulphate

Sulphate in known to inhibit anaerobic digestion at all levels as the sulphate reducing bacteria compete with the methanogenic bacteria for acetic acid and hydrogen as substrate. Research literature shows effects are difficult to assign to sulphate at levels in the leachate below 500 mg/l. Sulphate levels in all leachate samples from raw and test cell leachate were not detected (<2 mg/L). Verification of raw leachate analysis by Hills Laboratory gave a sulphate level of 7mg/L using ion chromatography (APHA 1 4110 B (modified) 23rd ed. 2017).

Gaseous hydrogen sulphide levels in the test cells are shown in Figure 38 and were found to be negligible. Except for test cell 3 and 5 which got as high as 146ppm.
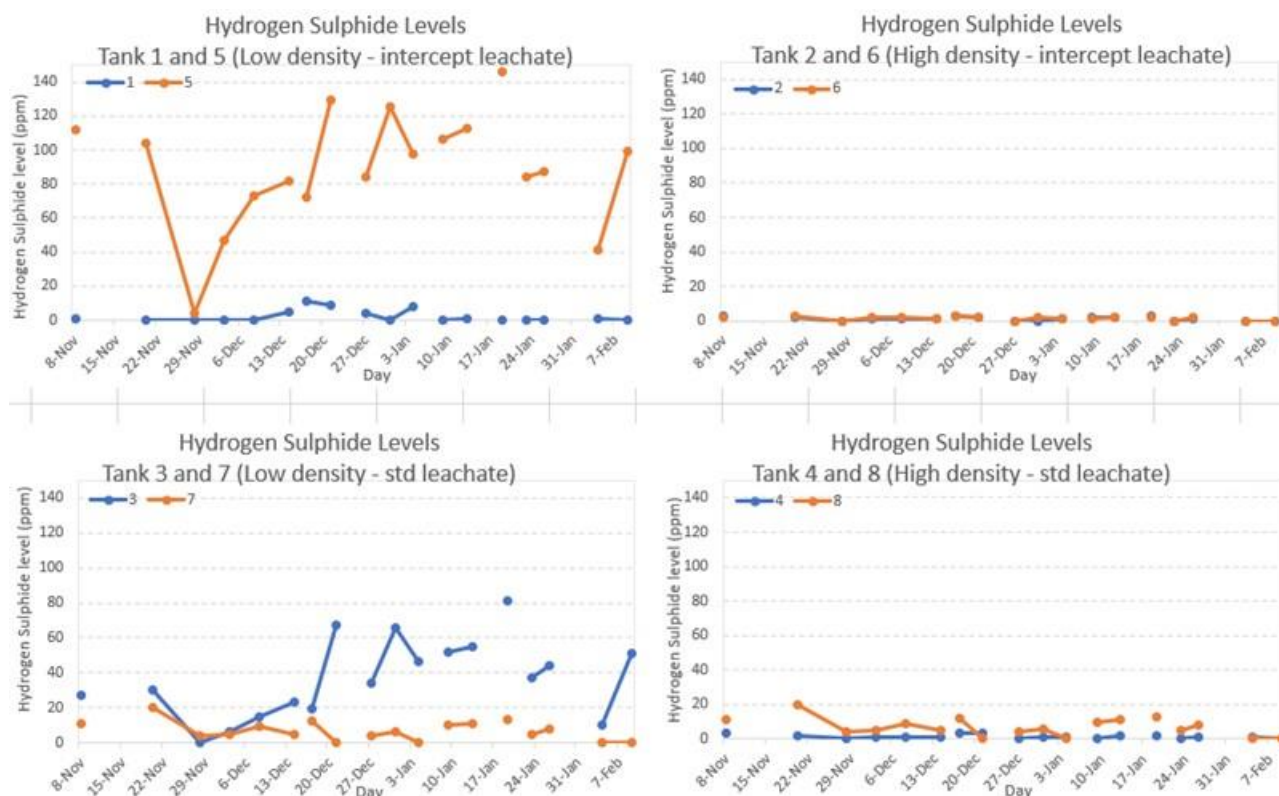
Figure 38 H_2S gas levels for all test cells (by duplicate)

The conclusion to be drawn from this is that test cells 3 and 5 may have experienced more reducing conditions.

Hydrogen sulphide gas levels recorded in the tube reactors were, like the test cells, also less than 150ppm for all tubes except in the earliest days of the pure rice samples (Figure 39).
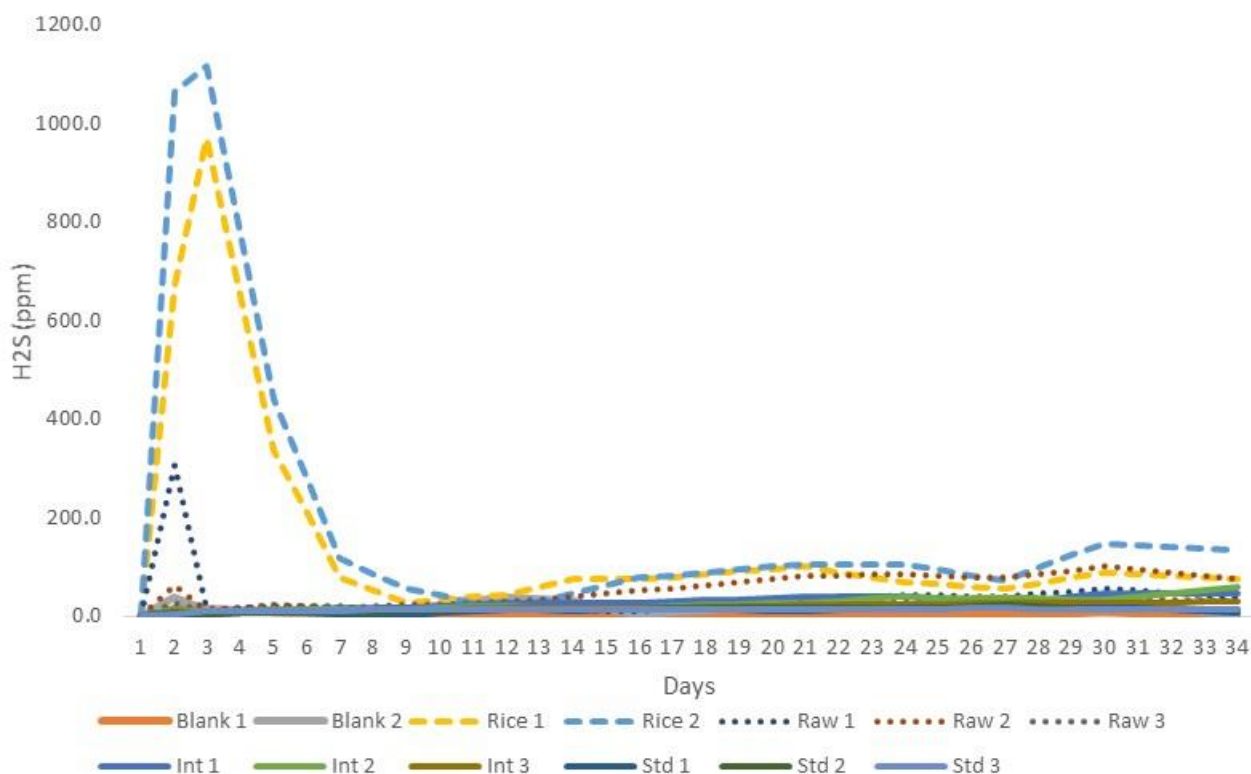
Figure 39 Hydrogen sulphide levels for tube reactors

Due to the low levels of sulphate in the leachate and low levels of hydrogen sulphide measured in the gas, sulphate is unlikely to be an inhibitory mechanism in the failure to produce gas in this experiment.

6.2.3. Ammonia

(Berge, Reinhart et al., 2005), Sponza and Ağdağ (2004), and Chung, Kim et al. (2015) state that the recirculation of leachate increases the rate of ammonification resulting in even higher levels of ammonia-nitrogen. Ammonia toxicity is the most common nitrogen problem with anaerobic digestion where ammonia can inhibit methane production at high concentrations. Zaman (2010) maintains that ammonia levels less than 1000 mg/L are a sign of a healthy reactor and quotes other research indicating that maintaining ammonia at or below 600 mg/L increases methane production and reduces the occurrence of ammonia toxicity. She also recognised that irrespective of pH, ammonia is toxic when above 3000 mg/L. The results of test cell ammonia levels in this experiment using Hach method never appeared higher than 695 mg/L. Total ammoniacal nitrogen in the raw leachate measured by Hills Laboratory using phenol/hypochlorite colourimetry (APHA 4500-NH3 H (modified) 23rd ed. 2017) was 920 mg/L.

If levels over 1000 mg/L are occurring inside the landfill this may be a concern with leachate recirculation at Kate Valley. Monitoring therefore becomes important. It is recommended that total

ammoniacal nitrogen be monitored regularly in recirculated leachate and in any raw leachate stored for future recirculation.

6.2.4. Oxygen

Oxygen is a known inhibitor of methanogenic activity. Sample collection did not follow the anaerobic protocol planned, with the samples exposed to air when dug up at Kate Valley prior to transfer to buckets for transport. There is a clear risk here of inhibited anaerobic recovery, as described by Loesche (1969). He maintains anaerobic recovery can be three to four times as successful for anaerobic bacteria manipulated in an anaerobic environment as those exposed to atmospheric conditions.

Anaerobic conditions are important as some anaerobic bacteria, known as strict anaerobic bacteria cannot tolerate oxygen levels above 0.5%. Strict anaerobic bacteria include some Clostridia and Treponema genera of which some species are known to be part of the taxonomic composition of bacteria identified in landfills by Sekhohola-Dlamini and Memory (2020). Pin-pointing specific anaerobic microbes that are both strict and landfill related is difficult given the plethora of microbes in landfill and knowledge to date. APPENDIX F – Microbial Diversity outlines the diversity and uncertainty around particularly anaerobic bacteria in landfills.

To ensure anaerobic conditions were optimised, once the samples were in the test cells, the oxygen levels in the headspace were constantly monitored. Results appear in Appendix D3 – Gas Results.

Oxygen is known to pass through HDPE membranes and Appendix C7 – Oxygen Transmission Rates Through HDPE outlines the approach taken to evaluate any potential risk using the EvalAmericas (2021) technical bulletin information on oxygen transmission rates. The potential increase in oxygen levels has been shown to be negligible and certainly less than the error of measurement. Rising oxygen levels over time would be a sign of possible permeation issues. It is recommended that the potential of oxygen transmission through any membranes be revisited if oxygen ingress is suspected.

There was unfortunately an error in the measurement of oxygen in the early stages of the experiment. The gas analyser appeared to be reading oxygen levels 3 to 4 percent higher than a recently calibrated meter. Methane levels were also noted as between 4 and 9 percent lower.
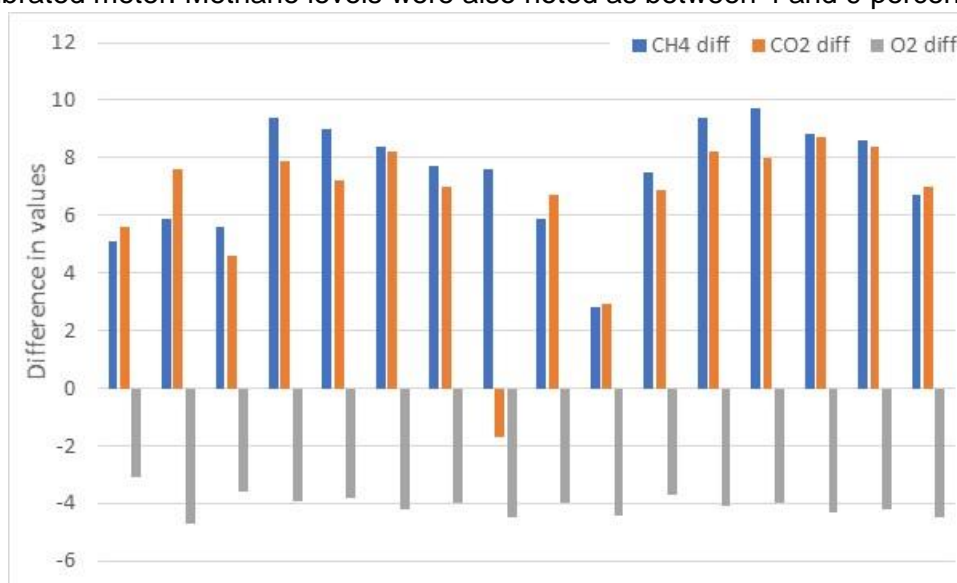
Figure 40 Gas analyser differences in gas levels

As a result of finding this erroneous meter it is anticipated that oxygen levels in the gas were not above 1% for all test cells and interestingly rarely above 0.5% for all low density cells (see Appendix D3 – Gas Results)

6.2.5. VFA Indicators

Zaman (2010) maintains having less than 1000 mg/L of total VFAs in leachate signifies a stable anaerobic digestion process which is supported by previous research. Values above these usually are signs of a stressed reactor. She further maintains that bioactivity could be inhibited if total VFAs exceed 6000 mg/L and reports other literature that did not observe methane when the TVFAs increased to over 13,000 mg/L. VFAs in leachate in this experiment was not detectable and certainly less than 10mg/L indicating that VFAs can not be causing an unhealthy reactor. However, VFAs are necessary for methanogenesis and the lack of them is concerning. Raw leachate levels of BOD were also very low (25mg/L). It raises the question of whether the issue is not what is inhibiting the methanogenesis but what is inhibiting the acidogenesis. The tube reactors experiments provide some indication that this is not the case as the refreshed waste tested delivered good biomethanation potential outcomes. The poor performance of the raw and spent waste in tube reactors does lead to some doubt. Continual measurement of VFA would help to investigate this further.

Regardless, initial indications point to a lack of organic material for the bacteria to metabolise. This question should be addressed in any future work and is certainly a necessary consideration for future experimental design.

6.2.6. Phosphate Availability

Phosphate is a required nutrient and can be inhibitory if at insufficient levels. Stoichiometric analysis can be a guide to phosphate levels required, but actual requirements are always much

higher. Phosphate levels in the leachate are of reactive orthophosphate. Levels in raw leachate are 12-30 times higher than those leaving the test cells indicating the phosphate is being utilised within the waste. Anaerobic biological species have an estimated stoichiometric requirement of 300:1:0.1 COD:N:P. Measured COD levels ranged between 8,500 to 35,000 mg/kg(dry) in the waste. The maximum theoretical stoichiometric phosphate requirement is approximately 12 mg/L in the leachate (if this was the source). The raw leachate had 32 mg/L which appears to be sufficient. However, leachate exiting the test cells contained between 1.07 and 2.64 mg/L phosphate, which indicates the phosphorous was being used and/or removed in some way.

The method for phosphate measurement states that interference occurs at ferrous levels greater than 100 mg/L. Ferrous levels recorded were less than 1 mg/L.

This casual analysis indicates that there is sufficient phosphorous in the raw leachate for metabolic requirements. Levels in the test cell leachates being significantly lower indicate that phosphorous is being utilised implying that for Kate Valley phosphorous may be a limiting factor in optimising biological degradation.

6.2.7. Chloride Content

Chloride is not a known inhibitor of methanogenesis, but its fate in landfills is not well understood. Chloride levels in the leachate leaving the test cells were around 1100-1200 mg/L, lower than the 1976 mg/L present in the raw leachate. This indicates chloride was being absorbed or reacting in the test cells.

6.3.　Organic Content

In line with postulations articulated by DeWalle (1978), that approximately 50% of potentially degradable organic material is protected in non-shredded waste in a landfill, it was expected that there would be sufficient organic material within the waste sampled, to support anaerobic activity for this experiment. This additional organic material would come from refuse lenses such as clumped waste particles, partially torn refuse bags or perhaps waste separated by structural voids as shown in Figure 41, freed during sampling protocols.

Figure 41 Partial torn plastic bag with structural void beneath

That this has not actually occurred is surprising, and further evidence was required to validate any conclusions drawn about availability of organic material, especially carbon sources.

Physically, the composition of the waste appears to have high levels of timber residue. This is not atypical of the MSW of developed countries (Zheng, Lü et al., 2015). Most of the timber apparent in the samples taken, especially in smaller size particles, appear to be fragments of plywood. This material is considered slowly biodegradable and is not expected to assist in the rapid methanogenic activity expected for this experiment.

Figure 42 5 mm retained waste particles showing timber fragments

Comparing the composition of the waste used in this experiment (Figure 21) with the New Zealand context in Figure 3 shows there is little organic content in larger portions of waste (the only sizes that could be clearly identified), especially readily biodegradable organic material (like grass, paper, leaves and food). Although the smaller sizing fractions (<2mm) could not be categorised they make up approximately 40% of the total waste (see Figure 43).
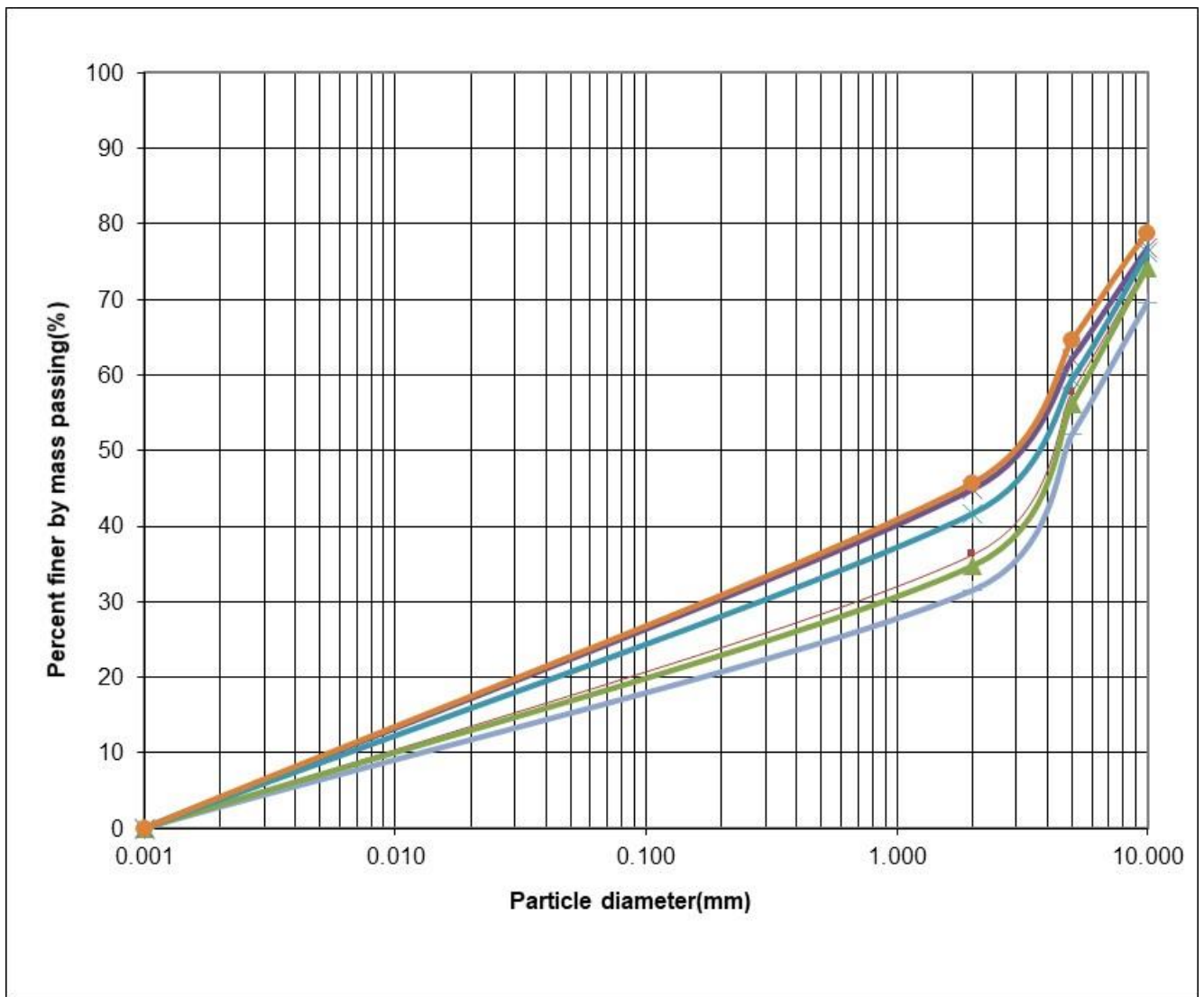
Figure 43 Particle size distribution <10 mm (dry sieved)

Conservatively, if it is assumed that all the volatile solids measured for the waste (Figure 24) are contained in the <2mm portion, then the maximum organic content is 15.3%. This is half of the organic content (30%) estimated by Perrot and Subiantoro (2018) for the New Zealand context as illustrated in Figure 3. Expanding Perrot and Subiantoro's classification to support all volatile solids by including timber, paper, nappies, textiles, and plastics with the organic stream (77.3% of NZ waste from Figure 3), enables a more credible volatile solids comparison of 15.3% to 77.3%. While not compelling, having only one fifth of the original volatile solids available, certainly emphasises the hypothesis that much of the highly biodegradable organics expected in the waste have been degraded. This is an important conclusion for the Kate Valley landfill of waste 10-years old and older.

In hindsight, measurement of organic content in the waste prior to the experiment may have enabled better anticipation of experimental outcomes and driven an expanded design parameter of using younger waste (5 years) as well as the 10-year old waste. The interpretation of literature sources led to overestimated biodegradable organic levels available in waste this age. It was only after the experiment had failed that it was clear that lower limits for completing methanogenesis described in the literature was valid for the waste sampled at Kate Valley.

It is worth noting at this point that landfill age does not govern leachate characteristics, rather it is the rate of stabilisation which is a function of the leachate processes. Landfill age is merely a convenient means, used by most researchers, of measuring changes in both leachate composition, gas flow and landfill stabilisation.

The analysis of volatile solids before and after the experiment show a loss of 0.35% (9.2% to 8.8%). Looking at the spread of data from 7.1% to 11.3% (standard deviation of 0.013) down to 6.8% to 10.2% (standard deviation of 0.012) respectively show this difference is difficult to validate statistically. However volatile solids does not distinguish between slowly biodegradable organics and readily degradable organics.

Huang, Wang et al. (2016) found in their leachate treatment experiments in anaerobic reactors that methanogenesis was only maintained with very high leachate COD levels (greater than 30,000 mg/L), as otherwise the landfill cell treating the leachate transited into the maturation phase. The levels of COD within this experiment were not anywhere near this level and BOD levels in the leachate were very low. Slowly biodegradable organics such as woody material and colloids are still present in this waste as residual organics. They do not degrade readily enough to support methanogenesis at rates for LFG production. Thus, two factors worth considering for failure of gas production are the impact of slowly biodegradable organics and the waste entering the maturation phase i.e. not being able to sustain methanogenesis measurable via COD and/or BOD levels in the leachate. Perhaps aligning COD half-life analysis to the research by Huang, Wang et al. (2016) and others such as Reinhart and Townsend (1998), may present new insights into this problem.

Distinction of slowly biodegradable organics from readily biodegradable organics in the waste is possible by measuring the biomethane potential of the waste samples in the test cells using an Anaerobic Respirometer developed at the University of Canterbury known as a tube batch reactor and described by Zaman (2010).

6.4.   Determination of the Biomethane Potential

Zaman (2010) suggests pH threshold values to indicate a healthy reactor include a pH between 6.9 – 8.0 (see section 2.4). Measured values for all tubes ranged between pH 7.1 and 7.7.

A methane composition of >50% indicates anaerobic activity and gas production had begun in the tube reactors. If the organic content is not available then methane levels will not reach this threshold. Levels <50% were observed in the blanks and all spent waste.

The results of the biomethanation potential experiments using tube reactors are presented in Appendix D5 – Biomethanation Potential and summarised in Table 20.

Table 20 Theoretical methane potential summary results

Tube contents	Average volume of gas produced (ml)	Maximum volume of gas produced (ml)
Blank	175	285
Rice	3415	4697
Raw waste	1037	2027
Spent waste	210	320
Refresh Kate Valley	5710	8198
Raw waste + rice	1898	2285
Refresh waste+ rice	9167	14038

This data suggests interesting aspects about the biomethanation potential of the waste used.

Firstly, there seems to be some organic material available in the waste for biomethanation. The waste material added was equivalent in organic material (as measured by volatile solids) to the rice, and this data indicates that it is possible that much of it represents slowly biodegradable organics.

The relatively high COD levels in the waste itself, up to 35,000 mg O_2/kg dry waste (Table 14), indicate potential for methane production. This is not reflected in leachate COD levels, which are relatively consistent but low,1,500 mg/L (Figure 33). Huang, Wang et al. (2016) suggests that low leachate COD may indicate a mature phase of landfill degradation where most of the organic material may be slowly biodegradable and measured as slowly biodegradable COD (sbCOD). The tube reactor data certainly does not run against that theory.

Secondly, the refreshed waste has a higher potential than the raw waste that had been stored for 6 months. This could be related to the heterogeneous nature of waste, or indicate that the waste stored had further biodegraded (despite being refrigerated). Figure 44 shows that the fresh waste has consistently higher methane levels than all the raw waste samples and Figure 45 shows volumes are significantly higher.
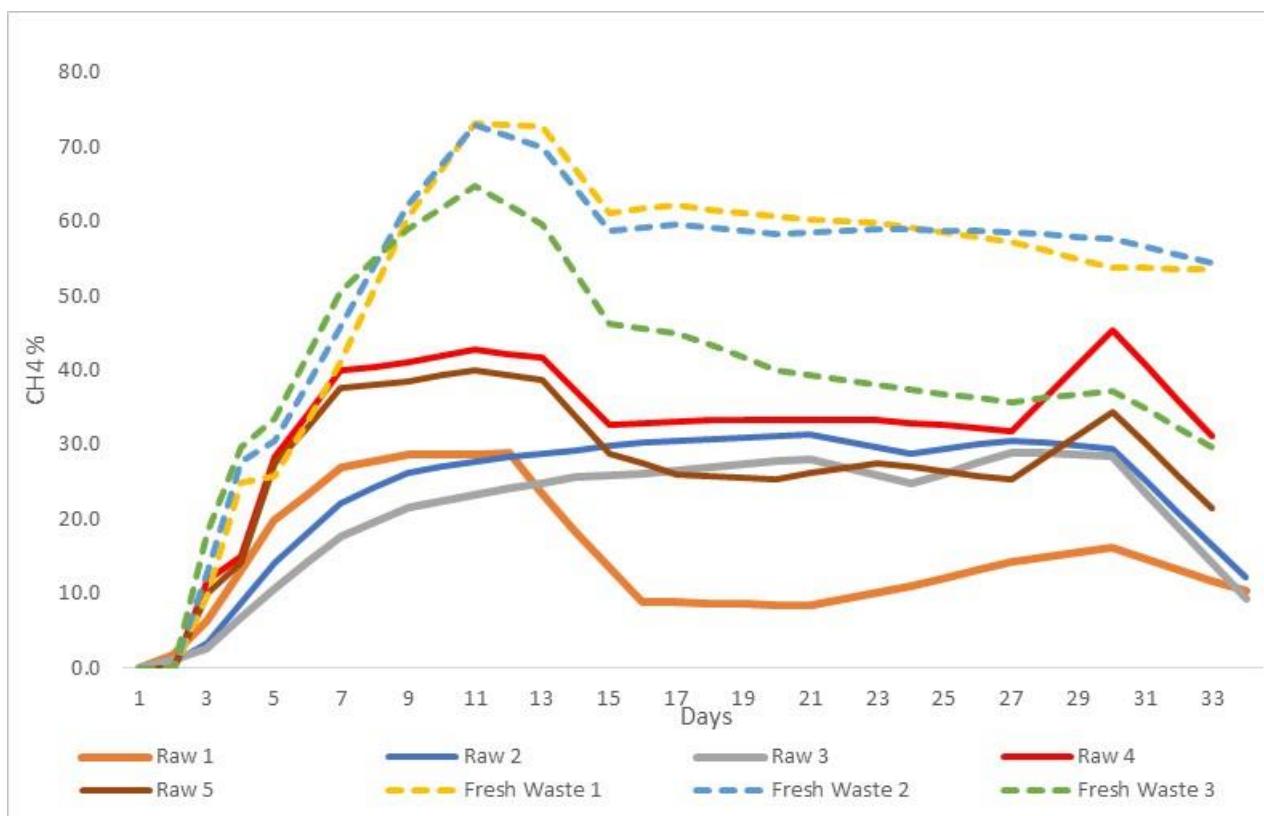
Figure 44 Raw and refreshed waste biomethanation potential – methane composition
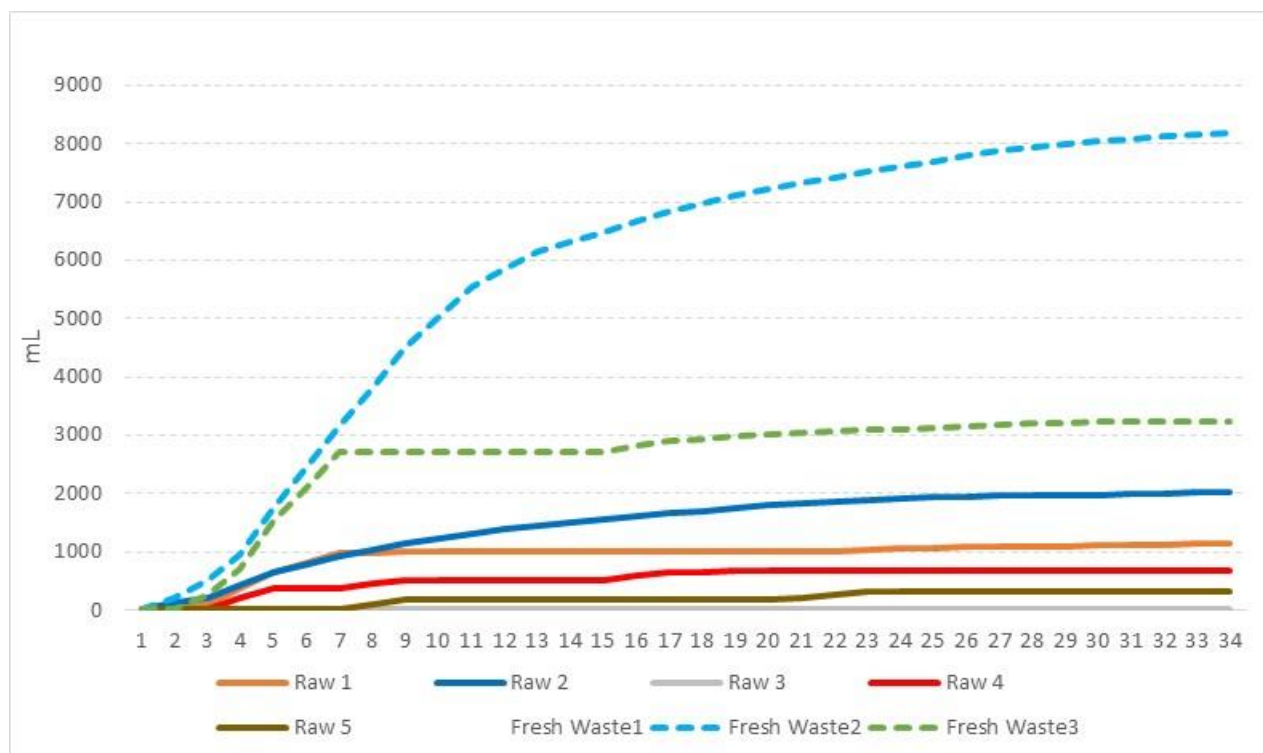
Figure 45 Raw and refreshed waste biomethanation potential – gas volume

Note the refreshed waste sample 1 did not record any gas volume and following the experiment it was discovered the µFlow flow meter was faulty.

The third aspect obvious from the biomethanation potential tube reactor experiment results is that assuming the raw waste did have some organic material with potential for methane production that this was used up somewhat in the original experiment run. All samples from the test cells recorded low methane concentrations (Figure 46) and low volumes (between 0 and 300 ml total) no better than the level of the blank (DSS only) samples.
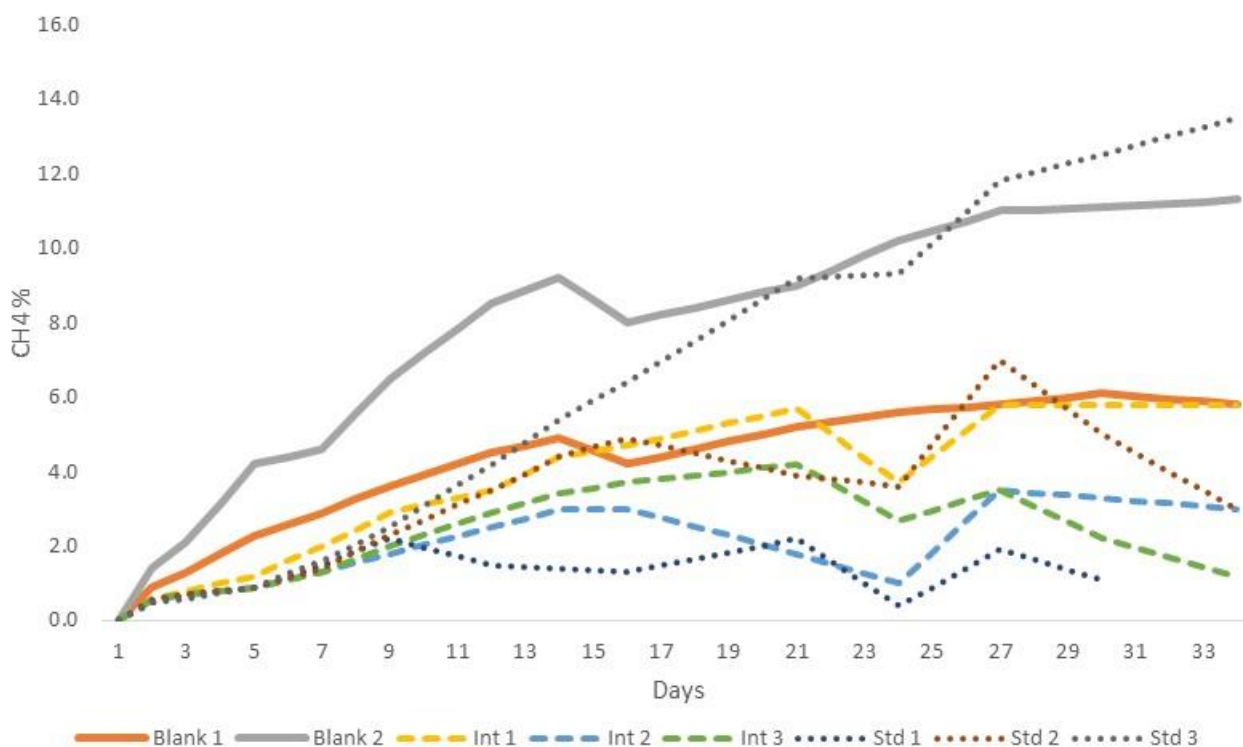
Figure 46 Blanks and Spent waste biomethanation potential – methane composition

This raises the question of why the spent waste did not produce gas in the test cells if it has less biomethanation potential than the raw waste following the test cell experiment. Although the test cell results did indicate there was methanogenic activity, as evidenced by the gas analysis in Figure 35, methane levels never got over 50%. This level of 50% is recognised by Zaman (2010) as a threshold for anaerobic activity and gas production for these tube reactors. It is also a threshold in the experience of the environmental laboratory for wastewater biodegradation. Perhaps it is also true for the solid waste in the test cells. This would mean that the anaerobic activity experienced was insufficient to produce gas in sufficient quantities in the test cells but sufficient to biodegrade any readily biodegradable organics present in the waste over the 163 days of the experiment.

Fourthly, additional organic material added in the form of rice to the raw and refreshed waste samples increased methane volumes. Table 21 compares measured volumes of gas from waste against equivalent measured volumes of gas from waste and rice.

Table 21 Average (Maximum) gas volume with and without rice component

Tube contents	Measured Waste without rice (ml)	Measured Waste and Rice (ml)	Calculated Waste less rice volume (ml)
Rice		3415 (4697)	0 (0)
Raw waste + rice	1037 (2027)	1898 (2285)	0 (0)
Refresh waste+ rice	5710 (8198)	9167 (14038)	5752 (9341)

There are some implications of the calculation in Table 21. For refreshed waste, the volumes reflect the cumulative impact of rice only and waste only tests. For the raw waste, while volumes increased, the effect of rice was subdued compared to the refreshed waste. This may indicate some form of inhibition. This inhibition may be caused by the storage that occurred or sub-sampling differences in loading the tube reactors. It is interesting that the raw waste performs worse than the refreshed waste in all aspects strongly suggesting storage has had an effect.

It was noted during the experiment that the method for measuring gas composition needs refining. The air in the line of the gas analyser was transferred to the first tube reactor during the first measurement. This phenomenon was apparent in the results of the tube reactors experiments as for the two sets of tube reactor experiments the first tube was consistently higher in oxygen.

Table 22 First tube in the series oxygen level comparison

Tube 1 oxygen level average	Average for set	Highest Tube 1	Next highest reading (not tube 1)
1.45%	0.36%	4.1%	3.6
8.21%	0.71%	12.6%	4.4

6.5. Leachate Changes

Landfills can act as an anaerobic digester for the leachate rather than the waste if leachate flow rates are high. A symptom of this behaviour is a lowering of biochemical oxygen demand (BOD) of the leachate over time. While BOD was not measured, Figure 34 shows this is not an obvious symptom experienced with COD in these test cells. It is recommended that BOD be measured to better interpret leachate recirculation flows.

However, the colour of the leachate does change as it passes through the waste columns. Figure 47 is an example of leachate colour that are consistent between test units. The left most sample of

Figure 47 (in the measuring cylinder) is typical of the low density test cells (top ones of the test unit). The measuring cylinder in the centre of the photo is indicative of the high density test cells (bottom ones of the test unit) and the beaker on the right holds the raw leachate sourced from Kate Valley with the waste. The difference in colour indicates some physical or chemical changes have occurred. Based on COD values (Figure 33 and Figure 34) it is likely to be due to mechanisms such as those described by Daniel (1993) and Qasim and Chiang (1994) outlined in APPENDIX E – Mechanisms of Chemical Movement in Leachate.
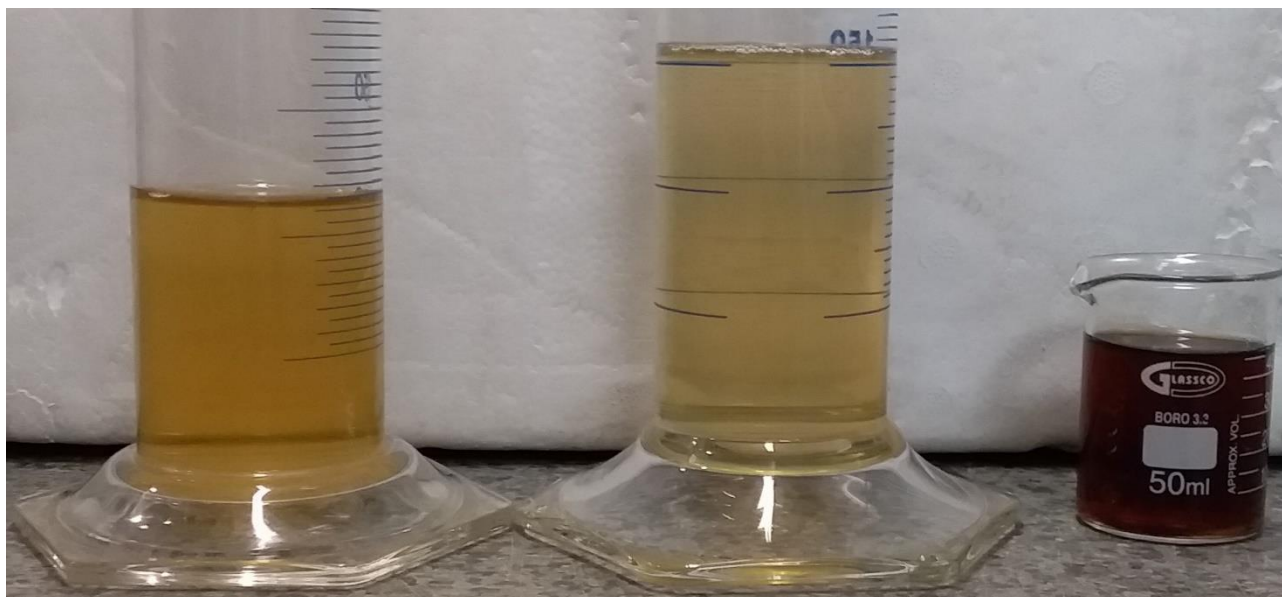
Figure 47 Low density, high density and raw leachate colours

An attempt was made to quantify the colour using the Hach spectrophotometer but substantial dilution was necessary to get the samples into range for colorimetric analysis. The output becomes meaningless with a diluted sample as outputs are reported in nanometres and are not proportional to dilution ratios.

Many reported laboratory scale experiments in landfill waste use a container with a sloping bottom. Altering the test cells using cement slurry, epoxy or other inert workable slurry to ensure that the tap is at the lowest point will ensure leachate volumes released are complete and more accurate than measured here.

6.6.   Sampling

6.6.1. Sampling Techniques

Anaerobic conditions were not maintained during the sampling of waste at Kate Valley. Maybe if the samples had been purged with nitrogen as soon as possible after sampling it would have helped flush out any oxygen and re-establish anaerobic conditions.

The sub-sampling that occurred in the laboratory for various tests is perceived to be successful. As discussed throughout this thesis the landfill and by inference waste itself is, as Senior (1995) described it, "*an exceptionally heterogeneous environment*" with organic molecules that are "*irregularly distributed in a medium composed of surfaces of varying nature and sporadically*

bathed in a fluid of uncertain composition." What this means is chemical and physical properties vary horizontally, vertically and temporally. To be able to take sub-samples and obtain consistent results for moisture content, field capacity, particle distribution and volatile solids when the material is even in smaller quantities still heterogenous, shows that the experimental design was valid and estimates made throughout this thesis have some reliability.

This data is not to be seen as representative of the Kate Valley landfill waste, as the samples have large pieces removed and the influence of these are unquantifiable, and known to be significant. Yet parallels should not be ignored.

6.6.2. Particle Sizing

Waste is considered a large particle sand, and as it degrades, more like a fine sand/silt. The particle sizing distribution shows that this assumption of waste is valid for the <20 mm distribution.

6.6.3. Moisture Content

Moisture content was quite consistent over 25 samples at 50-100 g. This indicates a consistent sub-sampling technique. However because of the small sample size, heterogeneity reflecting the landfill is probably compromised.

Given the moisture content of the waste samples was 23% with only one sample at 35% and the field capacity is a minimum of 31%, overall the waste at Kate Landfill can be considered not to be at field capacity. This was qualitatively verified on site during sampling, as described in section 3.3.2, where it was apparent that the soil-like particles were too dry to be sticky.

6.6.4. Field Capacity

The initial field capacity tests undertaken required large sample sizes to ensure a consistent material between samples and to allow for compaction testing. These samples were limited to three of each density division (high density and low density) and the field capacities measured were well spread over approximately a 9% range (Figure 23).

The initial tests of field capacity indicated that a loss of moisture was experienced with pressure. An organic content (humus) is usually thought to be responsible for this result, as soils with high organic content tend to have higher secondary soil consolidation (although this is not necessarily the only possible reason especially with heterogeneous waste).

Figure 48 shows the average leachate recirculation flow over the period for each test cell.
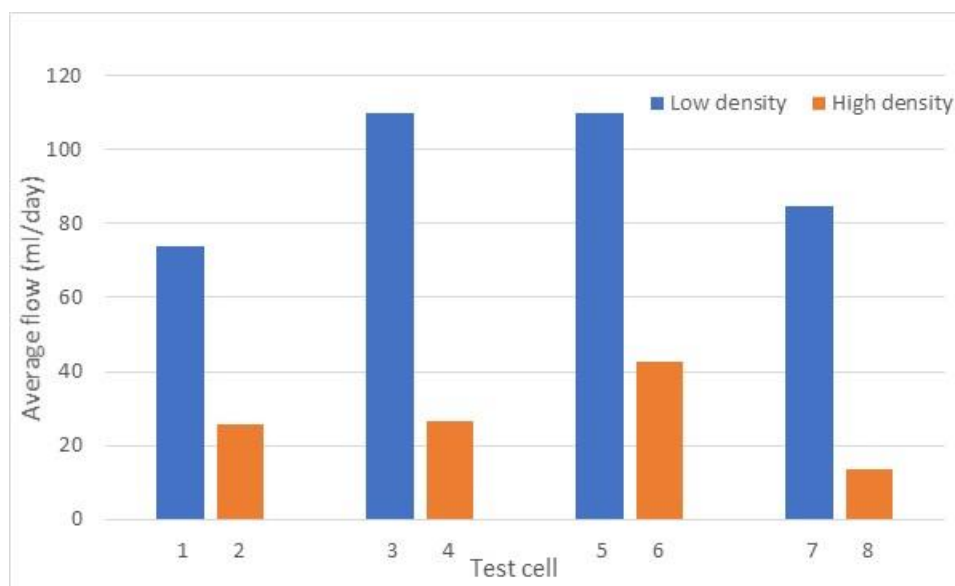
Figure 48 Average leachate recirculation flows (ml/day)

The difference in density can be seen to be affecting leachate flows quite markedly as expected.

The minimum volume apparent in the trends of leachate retained volumes given in Figure 30 and Figure 31 for each test cell could be considered an estimate of field capacity, as it is the lowest volume where no liquid flowed out of the test cell due to gravity. Table 23 summarise leachate addition and removal volumes and presents minimum retained leachate volumes for each test cell.

Table 23 Test cell leachate volume results

Leachate volume	Test cell							
	1	2	3	4	5	6	7	8
	Low density intercept	High density intercept	Low density std	High density std	Low density intercept	High density intercept	Low density std	High density std
Added (l)	10.60	10.15	12.39	10.29	12.65	9.25	12.47	9.08
Removed (l)	6.58	4.26	7.80	2.94	8.40	4.19	7.98	2.30
Minimum retained (l)	3.94	5.89	3.78	7.37	4.01	5.10	4.15	6.59

Charting this minimum retained volume, highlighting low and high density differences as shown in Figure 49, reveals an unexpected pattern. The high density, more highly compacted waste is holding more liquid than the low density waste. This may indicate that due to the continual recirculation of leachate using minimum long term volumes for estimating field capacity may not be allowing enough time for draining due to lower hydraulic conductivity of the more highly compacted waste.
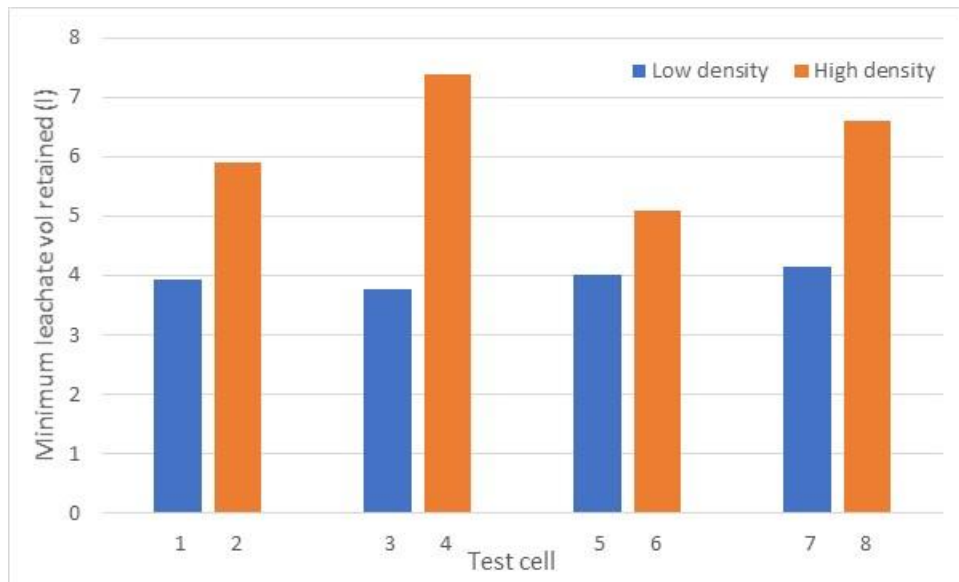
Figure 49 Minimum leachate volume retained by test cell highlighting density differences

Nonetheless estimating field capacity for each of the test cells based on these minimum retained leachate volumes over the course of the experiment and comparing this to initial estimates of field capacity appear in Table 24.

Table 24 Field capacity estimates

Field capacity parameter	Test cell							
	1	2	3	4	5	6	7	8
Type	Low density intercept	High density intercept	Low density std	High density std	Low density intercept	High density intercept	Low density std	High density std
Density (kg/m^3)	1007	1451	1003	1565	1066	1396	1121	1568
Initial estimated field capacity	38%	31%	38%	31%	38%	31%	38%	31%
Minimum leachate retained (l)	3.94	5.89	3.78	7.37	4.01	5.10	4.15	6.59
Test cell estimated field capacity	32%	32%	31%	33%	31%	31%	31%	32%

The difference in the field capacity estimated from retained leachate between the high and low density cells is not statistically significant with averages of 31% for low density test cells and 32% for high density test cells with a tight range despite varying densities. This is not surprising given the comments in section 2.8 about the outcome of hydraulic conductivity tests being both variable and difficult to attribute to any one specific characteristic. Adding to this the different hydraulic conductivities measured for the low density test cell 7 at 1.60x10^{-4} and the high density test cell 6 at 2.85x10^{-5} m/s it is apparent that this method of assessing field capacity requires more consideration including longer time for drainage of liquid and more samples.

A decrease in hydraulic conductivity with density was experienced. Density approaching 1600kg/m^3 experienced visible ponding (refer Table 13). It must be recognised that as waste is a highly heterogeneous material there is no definitive density at which hydraulic issues can be said to occur within a landfill. Fei and Zekkos (2013) outline settlement behaviour in terms of a systematic method to calculate the duration, strain, and long-term compression ratios. Investigation of this work may help to estimate when problems with low hydraulic conductivity may occur. That could also have implications for when or where to put in an intercept layer in active landfills.

6.6.5. Sampling Conclusion

The tightness of the range of compression data, moisture content, COD, and volatile solids (Figure 50, Figure 24 and Figure 25) gives confidence that the size of the experimental test cells and the techniques used for creating a sample to load the test cells, is effective and ensures consistency between and within samples and test cells.
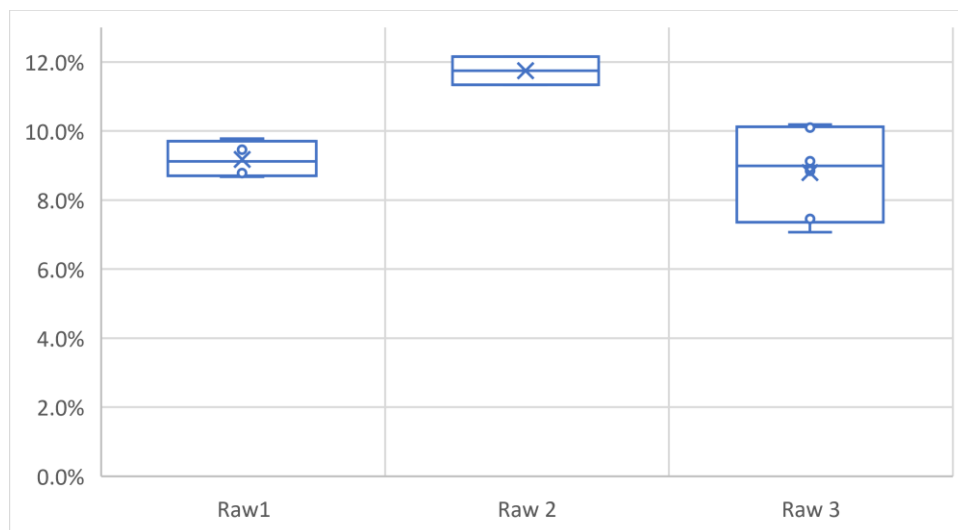
Figure 50 Volatile solids data spread (three sources and ten samples)

If sampling can on the whole be accepted as representative then it can be said that there appears to be no consistent or significant differences obvious between intercept leachate and standard leachate in any of the properties measured. The failure of the prime parameter of gas volumes has highlighted a lack of readily biodegradable organics. This is no doubt reflected in other parameters as COD, gas composition, nitrogen levels and VFA.

Differences were noted between the higher and lower density test cells but these are all attributable to existing known mechanisms discussed in the literature review (section 2.8). The exception to this is the volumes of leachate retained which appears to be attributable to hydraulic conductivity properties requiring significantly more time to ensure field capacity measurements are relevant. Also the heterogeneity of waste always offers a potential for both by-pass and localised hold-up of liquid within the waste matrix affecting hydraulic properties.

There was little pattern to be discerned between intercept leachate recirculation waste samples measured and the standard leachate recirculation samples.

7. CONCLUSIONS, IMPROVEMENTS AND FUTURE RESEARCH

This initial assessment of experimental design parameters was designed to observe differences between using intercepted landfill leachate for recirculation and base collected leachate, to answer the question "Can any benefit be measured in using leachate for recirculation from shallower depths compared to the bottom of a deep landfill?" The lack of gas production from the test cells impacted the outcomes, but while no observable differences were noted, successful outcomes were achieved.

The assessment of parameters to design and operate the experiment, collect representative waste samples and estimate recirculation volumes met with some success, and do not appear to have caused the lack of gas production in the experiment. The concepts outlined in the research design section (section 3.3) will help progress further research.

There is considerable evidence that the readily biodegradable organics expected in the waste sampled for this experiment were exhausted and inadequate for sustained methanogenesis. The experimental determination of the biomethane potential, measured volatile solids content, VFA concentrations in the leachate, and indications that the waste was entering the maturation stage, all support this conclusion.

Measurement of organic content in the waste prior to experimental setup would better anticipate outcomes. Testing biomethanation potential via tube anaerobic respirometer was an easy and relatively quick test to determine the availability of readily biodegradable organics. The reliance of literature sources herein, led to overestimation of biodegradability potential. It was only after the experiment had failed that it was clear that lower limits for the methanogenesis phase of biodegradation, as described in the literature, were valid for the waste sampled from Kate Valley.

The volatile solids measured were low and dominated by slowly biodegrading types such as wood and colloidal material. Differentiating between slowly biodegradable organics from readily biodegradable organics in the experiment by measurement of BOD, sbCOD and rbCOD or utilising tube reactor experiments may help to clarify if there is sufficient readily biodegradable organics for methanogenesis.

The lack of VFAs raises an interesting hypothesis that acidogenesis inhibition may be limiting the potential of the methanogenic bacteria. That question, coming at the end of the experiment, is a question to be addressed in any future work. Continual measurement of VFA and VFA species would help to investigate this.

There is also some indication that the waste may have passed the methanogenic stage. Low organic levels in leachate and falling gas levels are indicators that the waste may have been entering the maturation phase. Measurement of COD and/or BOD levels in the leachate, to monitor reducing rates of microbial activity, increasing nitrogen gas concentrations, and areas of high redox potential in the waste, as described in section 2.4 may assist in determining the phase of biodecomposition of the waste.

The tests developed to meet the original objectives of this research appear to have the potential to allow valid comparison of intercept leachate recirculation against bottom leachate recirculation. There are, however, improvements that could be made to better ensure success:

- Exclusion of oxygen when removing waste samples from Kate Valley and subsequently when transferring samples to test cells can only improve the confidence in having optimum conditions for the anaerobic bacteria to thrive.
- Density tests in 20 litre buckets were initially used to design the experiment. They were subsequently observed to not be representative of the 60 litre test cells. This suggests size and shape of the container influences density and therefore field capacity and hydraulic conductivity. Attention should be given to testing in the same containers as experimentation occurs. This approach is expected to provide more representative results, enabling better prediction of moisture addition and field capacity and avoiding such issues as the ponding experienced in some of the test cells.
- Using a test cell with a sloping bottom to ensure that the tap is at the lowest point will ensure leachate volumes measured are fully representative.
- Additional protocols for determining the gas composition of the tube reactors are required to ensure the air in the line of the gas analyser is not transferred to the first tube during the first measurement.

Finally, some useful information about the state of 10-year old waste at Kate Valley Landfill has been ascertained.

- There appears to be little organic material available for methanogenesis in 10-year old waste. This information can be used to test the sensitivity of the assumptions within the gas production models and may affect gas production longevity predictions.
- The waste at Kate Valley Landfill appears not to be at field capacity. This has implications for optimisation of gas production and addition of liquid. Up to 10% addition by volume seems likely to obtain field capacity.
- Phosphate may be a limiting nutrient in older waste at Kate Valley despite theoretical requirements indicating otherwise.
- Ammonia should be monitored along with pH to ensure healthy conditions remain for anaerobic degradation given the levels monitored within the experiment were approaching 1000mg/L.
- Waste is considered to be a large particle sand, and as it degrades is more like a fine sand/silt. The particle sizing distribution shows that this assumption is valid for the <20 mm portion of the waste.
- A decrease in hydraulic conductivity with density was experienced. Density approaching 1600kg/m^3 experienced visible ponding. It must be recognised that as waste is a highly heterogeneous material there is no definitive density at which hydraulic issues can be said to occur within a landfill.

8. GLOSSARY

It is recognised that much of the literature studied uses terminology that is either in common use or is not well defined. For example, the term leachate itself does not define whether it is synthetic, laboratory or landfill derived. Studies sometimes recognise this ambiguity such as the analysis by Reddy, Kumar et al. (2017) of mechanical models, but often, textbooks in particular, that rely on secondary sources summarising tertiary literature (Qasim and Chiang, 1994) are guilty of mixing information unnecessarily.

Landfill stabilisation is also a term with multiple meanings. It is used to refer to long-term stability of a landfill but can be referring to consistent leachate chemistry (Bolyard and Reinhart, 2016), mechanical strength of waste (Reddy, Kumar et al., 2017) or cessation of void collapse. Sometimes it refers generically to more than one of these properties (Chung, Kim et al., 2015). Thus, there is need to define the term stabilisation. This word encapsulates the idea that settling, decomposition, leaching of chemicals, and microbial activity has become minimised or non-existent and leachate quality is constant. It occurs at the end of Phase V of the biochemical representation explained in section 2.4.

Rather than rely on the reader's interpretation, the following definitions are supplied to:

- explain common acronyms
- define relevant industry terms
- differentiate or precisely define concepts as used within this thesis.

Term	Definition
Acronyms	
AS	Australian Standard
ASTM	Originally the acronym for American Society for Testing and Materials now an international standards organization
BOD$_5$	Biochemical Oxygen Demand (5 days)
CNRE	Civil and Natural Resources Engineering
COD	Chemical Oxygen Demand
DOC	Dissolved Organic Carbon
DSS	Digested Sewage Sludge
ELFM	Enhanced Landfill Mining
ISO	The International Organization for Standardization
ISWA	International Solid Waste Association
LCS	Leachate collection system
LFG	Landfill Gas
LRS	Leachate recirculation systems
MSW	Municipal Solid Waste
NZS	New Zealand Standards

rbCOD	readily biodegradable COD
sbCOD	slowly biodegradable COD
STP	Standard Temperature and Pressure
TDS	Total Dissolved Solids
TMP	Theoretical Methane Potential
TOC	Total Organic Carbon
TS	Total Solids
TSS	Total suspended solids
USEPA	United States Environmental Protection Agency
VFA	Volatile Fatty Acids
VS	Volatile Solids
WWTP	Wastewater Treatment Plant
Landfill specific terms	
Attenuation	The process of immobilising and retarding chemical constituents from moving with a liquid flow.
Bioreactor landfill	A controlled landfill where liquid and gas conditions are actively managed to accelerate or enhance biostabilisation of the waste.
C&D	Construction and Demolition waste.
Landfill*	A waste disposal site used for the controlled deposit of solid wastes onto or into land.
Landfill gas	A mixture of gases produced by the anaerobic degradation of solid waste. Predominantly methane it also contains carbon dioxide, ammonia, nitrogen, hydrogen, hydrogen sulphide, water vapour and trace amounts of other organic gases and vapours.
Leachate*	Liquid that, in passing through waste, extracts solutes, suspended solids and other components of the waste material through which it has passed. This includes liquid wastes themselves, liquid that drains as a result of waste compression, or the ongoing breakdown of organic matter.
Municipal Solid Waste*	Any non-hazardous, solid waste from household, commercial and/or industrial sources. It includes putrescible waste, garden waste, biosolids, and small amounts of treated industrial and clinical waste.
	It is recognised that municipal solid waste is likely to contain a small proportion of hazardous waste from households and small commercial premises that standard waste screening procedures will not detect. However, this quantity should not generally exceed 200 ml/tonne or 200 g/tonne.
Post-closure	The landfill in final shape after waste is no longer accepted. Can still be creating LFG and leachate, and settling.
Sanitary landfill	Landfill that is engineered to decrease surface decay, minimise odour, vermin and fire risk and improve aesthetics.

Solid waste	A waste that is not gaseous or liquid. May include sludges. Usually classified into MSW, C&D waste, and industrial waste.
Waste*	a) anything disposed of or discarded; and b) includes a type of waste that is defined by its composition or source (for example, organic waste, electronic waste, or construction and demolition waste); and c) to avoid doubt, includes any component or element of diverted material, if the component or element is disposed of or discarded.
WasteMINZ	A representative body of the waste and resource recovery sector in New Zealand.
Generic terms defined in this paper for clarity	
Field capacity	The maximum amount of moisture that can be retained by waste once downward gravity drainage has ceased. Measured as kg H_2O/kg wet refuse.
Grade A glassware	Laboratory glassware that has volumetric tolerances prescribed in ASTM E694 and has superior thermal and chemical resistance.
Grade B glassware	Laboratory glassware that has volumetric tolerance that is twice as large as Class "A". General purpose glassware that is not intended for prolonged exposure to chemicals.
Model	A mathematical representation of a relationship between two variables (typically for prediction or estimation).
Representation	A concept, analogy or interpretation that allows for understanding of complex processes.
Test cell	The containers used to experiment on the waste samples.
Test unit	Two test cells combined to simulate a landfill.
Tube reactor	An anaerobic respirometer developed and engineered by the CNRE Laboratory at the University of Canterbury.
Tedlar bags	Gas sampling bags made up of Polyvinyl Fluoride (PVF) film. A patented product of DuPont.

* WasteMINZ (2018)

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APPENDIX A – Summary of Benefits for Recirculation of Leachate

Abstract

Decades of research has highlighted fast waste stabilisation as key to landfill success. Waste stabilisation has been proven to be enhanced by the addition of moisture, buffer, anaerobic bacterial seed and leachate flow. Leachate recirculation supports enhancement of waste stabilisation as it can supply moisture, microbes and nutrients without additional water volumes being added to the landfill.

Removal of leachate from the base of a landfill mitigates the risk of harmful leachate leakage; recirculation of this leachate back into the landfill has been accepted practice to speed up the stabilisation process in landfills for several decades. Research using gas and leachate chemical parameters and landfill geophysical properties has quantified and validated the success of leachate recirculation in both laboratory tests and landfills worldwide. However leachate recirculation is limited by hydraulic balancing, chemical control of the recirculated leachate and lifecycle costs of equipment.

Context

Waste disposal to landfill is restricted by volume and by after-care requirements. Optimal lifecycle management for landfill owners is to make the best use of volume available, ensure post-closure risk is minimised and mitigated at optimum cost.

Landfills settle and stabilise over a long period, so hastening waste stabilisation is beneficial. McBean, Rovers et al. (1995) summarise the advantages of fast waste stabilisation as:
- degradation of waste occurs earlier, limiting the duration of the risk exposure
- methane gas production increases, improving gas utilisation efficiency
- leaching of contaminants happens in a shorter time
- leachate volumes and quality become predictable
- settlement creates airspace and releases additional volume
- the landfill moves quicker to its intended final use.

Christensen and Kjeldsen (1989) found that landfill degradation occurs faster by promoting anaerobic biological activity. Key to this is supplying water and nutrients. McBean, Rovers et al. (1995) note four ways that enhancement of waste stabilisation happens:
- the addition of moisture
- buffer to control pH
- anaerobic bacterial seed
- leachate flow.

Moisture

Laboratory test cells, pilot-scale studies and full scale experiences like those listed by Reinhart and Townsend (1998) show that control of moisture enhances stabilisation of waste. When the moisture content of the waste exceeds capacity, free-flowing liquid flushes and rinses waste and

acts as a transport mechanism for nutrients, microbes, and chemical inhibitors of microbes to other areas within the landfill (McBean, Rovers et al., 1995). According to Zeiss (1992), the heterogeneity of landfills has a strong influence on hydrology and as such, flushing is often limited, as preferential pathways exist.

Ko, Yang et al. (2016) and Bolyard and Reinhart (2016) support flushing as one of the functions served by leachate movement. Water alone was shown to serve the same purpose in a similar manner. The addition of any moisture is known to improve key biological reactions, however, leachate has the advantage of supplying microbes and nutrients. This leads to increased anaerobic decomposition rates in a complex interrelated fashion beyond that of just using water. Additionally, avoiding the addition of superfluous water decreases the volume of leachate that will ultimately need to be treated.

Leachate Recirculation

Reddy, Kumar et al. (2017) describe how leachate recirculation results in improved moisture distribution and faster microbiological activity which leads to accelerated waste stabilisation. Fei and Zekkos (2013) also note that liquid addition and/or leachate recirculation promotes waste biodegradation, by enhancing microbial degradation, specifically methanogenic activity (the primary process for degradation) through aspects such as:

- raising moisture levels in the waste to allow moisture movement
- supplying water and other nutrients
- flushing
- minimising some inhibiting chemicals (although there are risks with other inhibiting chemicals such as pH, ammonia/ammonium).

The recirculation of leachate back into the landfill has been accepted practice for several decades now (Pohland, 1975), and is known to speed up the stabilisation process as well as even out hydraulic peaks and chemical concentrations in leachate to be treated. Advantages and disadvantages of leachate recirculation are outlined in Table 25.

Table 25 Advantages and disadvantages of leachate recirculation

Increased methane production

Improved flushing and rinsing of waste

Improved chemical, nutrient and water transport

Enhanced biological activity leading to faster waste degradation

Buffered leachate volume

Delayed leachate disposal

Treatment for leachate organics

Reduction of leachate volumes through evaporation and sorption

Increased solids in leachate collection equipment

Increased bioclogging

Odours

Capital and maintenance costs of equipment

Necessity to handle higher hydraulic loads

Control of leachate chemistry

Increased risk of anaerobic disruption due to pH, ammonia/ammonium, toxin levels in leachate

Sources: Townsend (2015), Chung, Kim et al. (2015), Sponza and Ağdağ (2004), McBean, Rovers et al. (1995), Qasim and Chiang (1994), Leuschner (1989) and Pohland (1975)

Leachate recirculation is known to reduce organic content of waste most often measured as Biochemical Oxygen Demand (BOD) or Chemical Oxygen Demand (COD). Reinhart and Townsend (1998) for example show, using studies of COD half-lives, that COD levels are reduced by five to ten times (30,000 to less than 3,000 mg/L) within short periods (5 years) using recirculated leachate in closed landfills. This is based on their own and previous research by others in both laboratory studies and landfill data.

In summary recirculating leachate hastens waste stabilisation through enhancing anaerobic degradation. Christensen, Cossu et al. (1989) reported that from analysis done on 15 operating landfills, recirculating leachate achieved an equivalent level of degradation to layering the waste thinly, at 0.5 -1 m lifts.

Measuring Degradation

Leachate recirculation experiments typically track changes to chemistry, biological activity and waste settlement. These form part of the basis of our understanding of the complex and interrelated processes that occur in landfills.

Various models have been used to provide insight into the microbial and chemical degradation processes that are occurring inside a landfill. Stanforth, Ham et al. (1979) and Ehrig (1983), studied leachate quality during landfill degradation. Their models describe the microbial activity

responsible for waste degradation which can be summarised as a short aerobic phase, followed by fermentative anaerobic bacteria responsible for high volatile fatty acid concentrations, followed by the rise of methanogenic anaerobic degradation and methane production. Each of these phases decompose organic materials and degrade some inorganic components in the waste and can be measured as changes in leachate quality over time. It is known that for most of the life of a landfill methanogenic anaerobic conditions dominate.

This type of modelling allowed Ehrig and others following him to use gas production and leachate chemical parameters to better define the degradation of waste. The various phases can be understood in terms of appearance of, or changes to, relative concentrations of compounds such as water. nitrogen, oxygen, methane, carbon dioxide, hydrogen, volatile fatty acids, ammonia, sulphate, and metals. Other parameters such as pH, alkalinity, redox potential, COD/BOD, and metal precipitate rates also support the interpretation and development of degradation progress.

The analysis of parameters such as these allow the degradation occurring to be understood despite the differences within a landfill and between landfills.

Uncertainty

Understanding the processes occurring in landfills is necessary to improve their operation and ensure long-term risks are minimised. It is clear though, that landfills are highly variable in many aspects. Gettinby, Sarsby et al. (1996) found highly variable leachate composition in published data caused by many factors including maturity of waste degradation, non-standardised sampling techniques and non-homogenous nature of landfills. Their summarised general trends on leachate quality cuts through some of the inconsistency in published data while acknowledging high variability as a problem with literature on solid waste. Qasim and Chiang (1994) summarise analyte information from 12 other researchers, some had sampled up to 30 landfills, and concluded that there is large variation in many of the leachate quality properties. For example, COD levels vary from 3,000 mg/L to 60,000 mg/L for leachate in landfills less than 2 years old. Interestingly while many analytes vary by up to 2 orders of magnitude in landfills less than 2 years old the range is much reduced in mature landfills (for example COD of between only 100-500 mg/L in the same study).

Leachate Quality Control

It is known that leachate, if continually recirculated without treatment or chemical modification, can interfere with and even stop bioactivity. The causes of this vary with the type of biological activity present in the waste but are known to include lower pH, an increase in toxic chemicals, particularly ammonia, and reduced availability of organic material due to redox conditions. Bacterial activity is also exposed to a variety of abiotic factors in the highly heterogeneous landfill system. McBean, Rovers et al. (1995) report that for methane forming bacteria the variability in oxygen, hydrogen, pH/alkalinity, sulphate, nutrients, inhibitors, temperature and water content affects the ecosystem and consequently the rate of activity. It can be seen then that control of leachate quality during recirculation may be necessary to ensure positive biodegradation occurs.

Experience in leachate recirculation has shown that except for arid regions or during dry spells, excess leachate is typically produced. While this can be retained in the landfill to some extent,

leakage and structural integrity risk minimisation strategies demand keeping leachate head at the base of the landfill to a minimum. It is therefore normal practice as part of leachate volume and quality control, to bleed off collected leachate for treatment or disposal.

However, it is worth mentioning that leachate recirculation is also a partial treatment process for leachate, as the landfill can be considered a type of anaerobic digester that can decrease organic carbon content and has some capacity to sorb toxins.

Ultimately, excess recirculated leachate will also need to be treated.

Leachate Flow Control

Degueurce, Trémier et al. (2016) showed that for solid state anaerobic reactors control of recirculating leachate flow is important for optimising methane production. They showed that both flow rate (the volume applied) and the frequency it is applied are important factors. This is a commonly understood phenomenon (consider watering a garden) and not an unexpected result. It is reasonable to expect that it is also valid for landfills. Feng, Bai et al. (2018) explain that while high flow is needed to maximise settlement of landfill, this gives rise to a higher recirculating load in the leachate collection system, which is more expensive and higher risk.

Ongoing Challenges

The heterogeneous nature of waste inevitably allows the bypass and short-circuiting of liquids, additionally waste is often present in bags or containers which excludes liquids. This means there are certainly localised areas of the waste within the landfill that do not participate in leachate recirculation mechanisms and do not degrade at the enhanced rate. This can lead to gas being produced over a longer period of time than that anticipated by analysis of enhanced degradation.

Some researchers have recommended and verified leachate recirculation rates, but these rates differ depending on waste and landfill characteristics. While initial estimates may be useful for a starting position, ultimate control is site specific and varies over time.

Some aspects of biological and chemical effects brought about by leachate recirculation are well understood but there are still aspects that require research. The ultimate fate of all chemicals in a landfill is not understood and less so the effect leachate recirculation has on this. In parallel while leachate recirculation is known to be beneficial for methanogenic decomposition the fate of other microbes playing a part in the breakdown of other waste for example fungi breaking down lignin-bound cellulose is not well understood.

Health and Safety Considerations

The health, safety and environmental risks in the experimental are inherent in the production of gas, the quantity of waste to be used and the quality of the leachate. An initial risk assessment completed is summarised in Table 26. As the laboratory continually has anaerobic digestion experiments, current risk controls are based on current laboratory procedures and protocols. Further mitigation involves specific experimental control.

Table 26 Initial risk assessment of proposed experiment

Hazard	Risk rating	Current controls	Mitigation
Heavy lifting	16	Move waste incrementally, setup experiment empty	Look for alternatives
Height	12	Framework used. Steps are solid	
Gas leak	9	Experience in lab, room vented for experiments	Soap check when experiment starts
Dermal contact with waste	9	PPE	
Dermal contact with leachate	9	PPE	
Interference	9	Laboratory rules	Hazardous labelling of test cells
Gas explosion	8	Experience in lab, vent, ignition sources minimal	Remove ignition sources close by
Excessive gas produced	6	Experience in lab, limited size of test cell	Monitor flow carefully during initial stages
Leachate spillage	6	Controlled Lab env, Clean up, PPE	Care
Waste spillage	6	Controlled Lab env, Clean up, PPE	Care

Chemical Oxygen Demand COD

Based on APHA Standard Methods Section 5220. The samples are prepared in the fume cupboard with the sink. Samples are digested at 150°C for 2 hrs, cooled and read at 620 nm on the Hach Spectrophotometer.

Take Dragon blood and COD Quality Control (QC) (KHP standard) out of the fridge. Put dragon blood on a magnetic stirrer. Pour some of the QC (usually about 10 ml) in a beaker and seal with wrap to ensure it is not contaminated. Leave them to warm up to room temperature (at least 30 min). Return the QC standard bottle back in the fridge.

Turn the Hach digestion machine on, to COD digestion (150ºC).

Prepare on a metallic rack COD tubes with 1 blank, 2 QCs and 2 of each sample. The test is for soluble COD. The samples of leachate were filtered to ensure consistency and diluted 1:1 with Deionised water.

Add 5 ml of dragon blood to each tube. Check levels are roughly the same. Add 2 ml of Deionised water to the blank, 2 ml of QC solution to the QCs, and 2 ml of sample to each corresponding tube. Close the lids very tightly. DO NOT SHAKE at this stage.

Bring the rack of tubes to the Hach digestion machine. When it has finished warming up (it rings once), take 2 tubes at a time, give them an inverted 2 shakes and place in the machine. Repeat for all tubes. Then push "start" on the machine and the digestion process starts for 2 hours. It will ring when the 2 hours are done. Place the tubes on the rack to cool down for 30 min. Go the Hach spectrophotometer and to "User programs" to read the results of COD in mg/L. Press start.

Waste COD goes into the waste COD bottles (to be disposed of correctly by Chemwaste). Tubes only need rinsing with tap water and DI water.

Geotech Tests

As the experiment is running in an anaerobic environment, the low density and high density samples necessary for geotechnical evaluation are generated using the same techniques used to fill the test cells rather than sample from the test cells themselves.

As waste is heterogeneous in nature it is impossible to obtain identical densities and even sampling the same experiment would lead to a range of outcomes. Therefore it is intended that the geotechnical outputs be presented as a range using multiple samples to enable the experimental outputs to be interpolated within these ranges.

Sampling of 4x 20 litre buckets of waste from Kate Valley landfill available will be performed by core sampling using a two-inch plastic pipe through each of four buckets and mixing the resultant samples into one combined sample. This combined sample can then be cut by standard soil sampling practices to perform the various tests. The coring, mixing and cutting can be repeated as many times as necessary to perform all tests required.

APPENDIX C – Design Calculations

Appendix C1 – Test Cell Dimensions

As the experiment is designed to simulate a landfill and incorporate two test cells making up a test unit it is preferred if the two test cells were stacked one on top of the other with room above and beneath each test call to access gas and leachate. Figure 51 is a sketch of the wall elevation of the controlled environment laboratory (2.5 m by 3 m) and the likely configuration and therefore dimensions of the test cells.
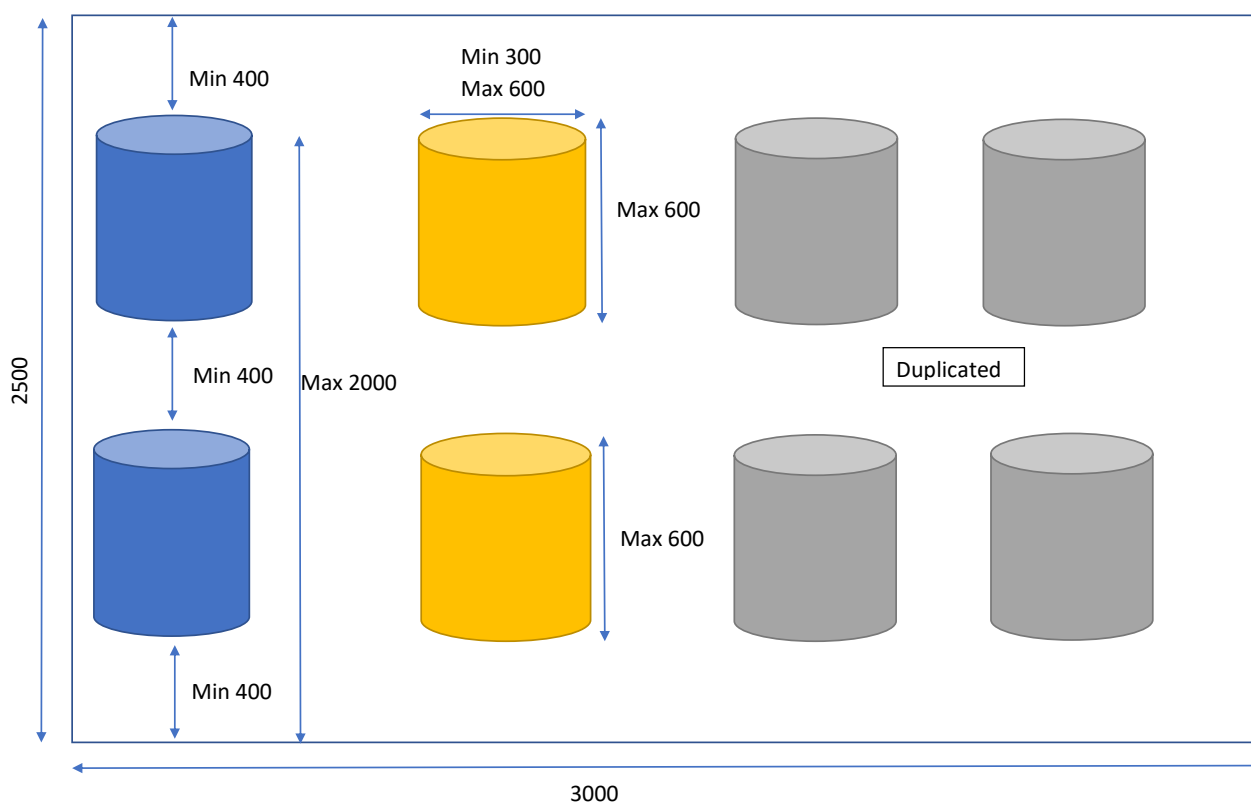

Figure 51 Configuration and space availability for test cells

60 litre (nominal) polyethylene off the shelf barrels were sourced from Stowers Containment Solutions in Christchurch that fulfilled these space criteria best without the need to custom build. The test cells were housed in a Dexion® steel framed rack in the laboratory (see section 3.3.7 for photo).

The internal dimensions of the 60 litre barrel are 400 mm diameter by 620 mm high. The barrel is tapered top and bottom. Because of the position and specifically the height of the spigot at the bottom of the barrel the drainage layer was designed at 90 mm of gravel and 65 mm of sand. Given the short tenure of the testing and small volumes of leachate expected, 360º garden micro-sprayers were used to disperse leachate over the top of the waste. The microspray nipple and tube was placed in the centre of the lid and tested at various heights above the waste to ensure the best coverage of spray across the barrel diameter without excess running down the sides. Using the peristaltic pump set at 2.5

Figure 52 Dimensions of test cell showing nominal drainage and headspace

(flow rate=0.125 l/min), this height was found to be 90 mm off the waste. Installation of the microspray unit required 40 mm down from lid giving 130 mm headspace in total. For small volumes (less than 60 mm) a syringe could be used to return the leachate as it was found that maximum hand compression of the syringe mimicked the pump.

The bottom of each test cell needed a 90 mm drainage layer of gravel to adequately cover the spigot to which an additional 50 mm of sand with geotextile was added to ensure filtration and protection of the drainage layer. Given the barrel height is 620 mm these dimensions define the waste depth to 350 mm or 37 litres.

Appendix C2 – Field Capacity

Estimates of leachate volumes can be obtained from other research in the literature using assumptions around initial moisture content of the waste, field capacity, density and test cell dimensions. Esteban-Altabella, Colomer-Mendoza et al. (2017) found major differences between density (ρ) and field capacity (FC) in refuse due to their composition, degrees of decomposition, and level of compaction. They outline a mathematical relationship to calculate the relationship between the field capacity and density of waste. According to their data a strong negative correlation was found between density and field capacity (r = 0.828).

Their result fits within the interval found within literature of real landfills, large pilot studies and laboratory tests. The best fit linear regression line is given as FC = - 0.699$\rho_{(dry)}$ + 1.142. Note density is dry weight and the abscissa is in error (should be tonne/m^3).
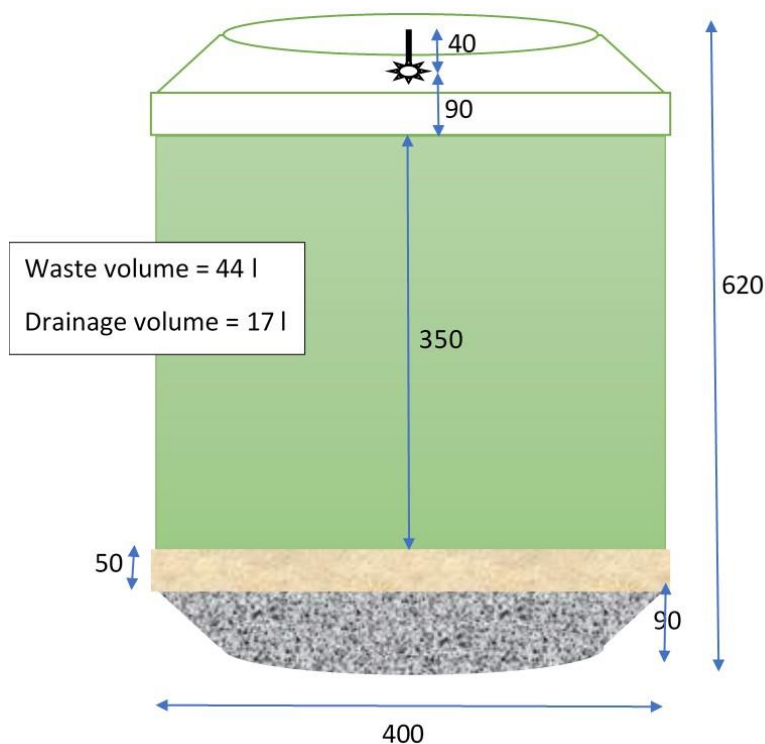
Their expectation of field capacity is less than 40% (by wet weight) and ranges from 400-700 litres of landfill leachate per tonne of waste (this sits within a range of 20-40% for landfill as given by Beaven (1995)).

Tolaymat, Kim et al. (2013) also compared density to field capacity but used wet density. They found the relationship between dry density and field capacity from other researchers' data could be best described as an exponential relationship:

$$FC = 0.267 + 0.652e^{-0.0024\rho}$$

The USEPA's commonly used leachate generation model, the hydrologic evaluation of landfill performance (HELP), provides a default volumetric field capacity of 29.2% (35.1% by wet weight, assuming a waste bulk density of 831 kg/m³). This is lower than both the estimates given above but sets an established lower bound.

Canziani and Cossu (1989) also report field capacity ranges against density although at significantly lower density ranges. These report within the data anticipated by Tolaymat, Kim et al. (2013) and Esteban-Altabella, Colomer-Mendoza et al. (2017). Figure 53 shows these three datasets together for comparison (converted to a wet basis).
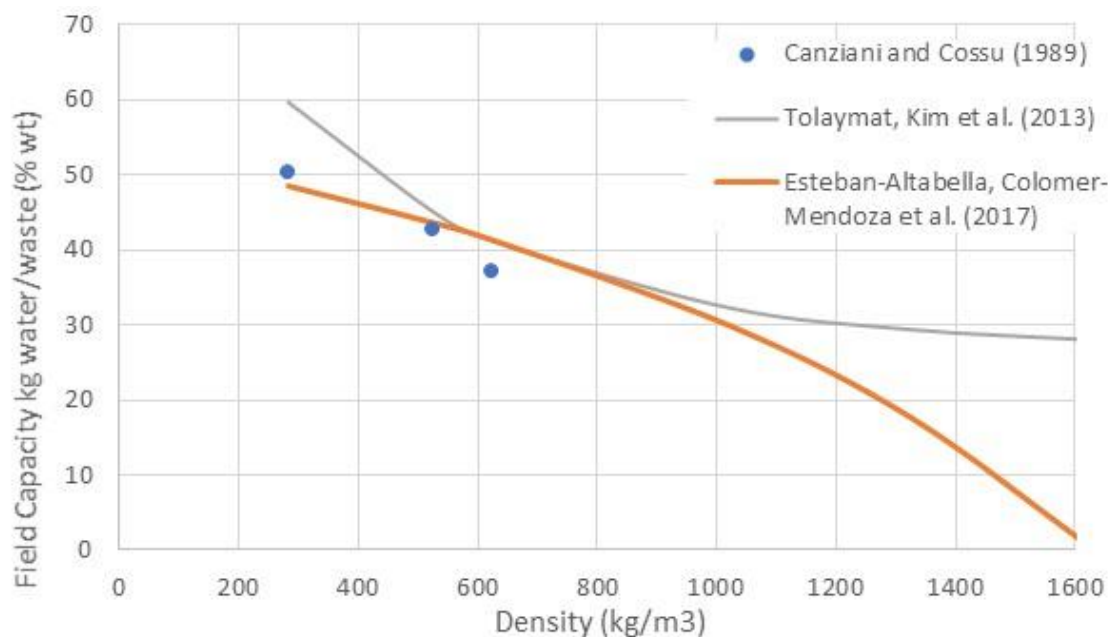

Figure 53 Density to field capacity relationship comparison (wet waste basis)

At typical densities assumed from section 2.8 (minimum 1020 kg/m³ at the surface to maximum 1632 kg/m³ at 30 m deep) field capacity can be estimated as shown in Table 27. Note some researchers measure field capacity on a dry weight basis (Esteban-Altabella, Colomer-Mendoza et al., 2017) and some on a wet weight basis (Tolaymat, Kim et al., 2013). For consistency this thesis uses a wet weight basis to avoid unnecessary assumptions of waste moisture content. Where conversion is required, moisture content is assumed to be 23%.

Table 27 Field capacity estimates for design

Source	Basis	Field capacity low density (1020 kg/m³)	Field capacity high density (1632 kg/m³)
Esteban-Altabella, Colomer-Mendoza et al. (2017)	FC = --0.699$\rho_{(dry)}$ + 1.142 dry weight basis	30%	N/A
Tolaymat, Kim et al. (2013)	FC= 0.267 + 0.652e$^{-0.0024\rho}$ wet weight basis <40% by wet weight	32% <40%	28% <40%
Qasim and Chiang (1994)	between 24% and 31%	31%	24%
USEPA HELP model	35.1% FC at 831 kg/m³	<35.1%	<35.1%

Qian, Koerner et al. (2002) assert that field capacity is a difficult parameter to assess because of the highly variable nature of waste in landfills. Conservative field capacity is used for design to allow confidence that liquid requirements will be less. Thus, a field capacity of 40% was the original design criteria to estimate leachate quantities required to be collected. Section 5.1 gives actual field capacity measured from waste samples collected and the field capacity of the final experimental test cells.

Appendix C3 – Moisture Addition

Moisture addition rates to achieve field capacity targets can be simply determined by considering initial moisture content of the waste. The volume of leachate (V) required to bring waste from an initial moisture content (M_i; % wet weight) to a target moisture content (M_t; % wet weight, assumed for design to be 40%) may be calculated from

$$V = \frac{M_t - M_i}{100 - M_t} C$$

where C is a conversion factor of 1 litre/kg. Figure 54 adapted from Tolaymat, Kim et al. (2013) applies this relationship to allow an initial estimate of volume of leachate required.
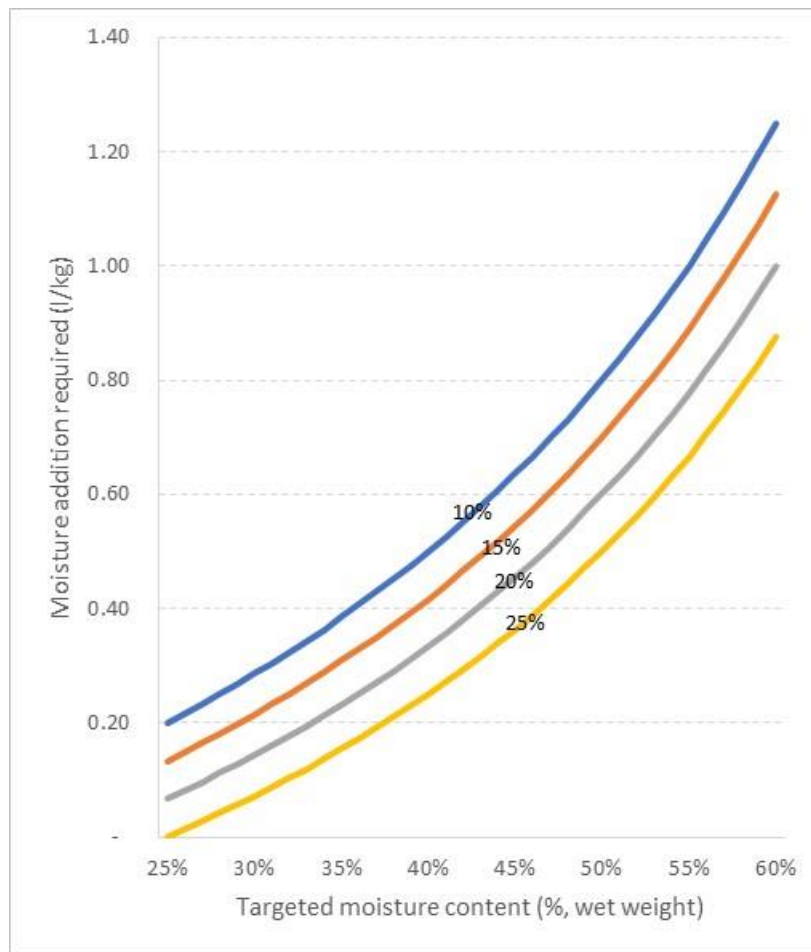

Figure 54 Moisture-addition requirements as a function of target moisture content
Reproduced from: Tolaymat, Kim et al. (2013)

From this leachate volumes can be calculated assuming an initial moisture value. Section 2.8.3 indicates initial moisture content range to be expected as 20-35%. Assuming a minimum 20% and a target field capacity of 40% gives a moisture addition necessary of 0.33 litres per kg. Assuming a density of 1020 kg/m^3 for the top barrels and 1632 kg/m^3 for the bottom barrels allowing room for headspace and drainage (as shown in Figure 52 and described in section 3.3) gives a maximum total leachate requirement of 185 litres to achieve field capacity.

Appendix C4 – Leachate Recirculation

From a variety of research, recirculation levels vary from ml to m^3 even within the same text reporting other researchers results (Reinhart and Townsend (1998)) , with some researchers even just reporting percentage volume e.g. Sponza and Ağdağ (2004) reports 13% of reactor volume per day. There is a balance however in supplying nutrients via the leachate recirculated: Too much leachate recirculation has the potential to see the bacteria using the nutrients in the leachate rather than in the waste as source of nutrients, turning the landfill into a leachate treatment facility and slowing down waste stabilisation. Insufficient flow interferes with the optimal conditions for microbial activity.

Table 28 give some indication of the ranges available in the literature crudely converted to the equivalent for the test cells here (mass, area or volume proportional).

Table 28 Recirculation volumes from literature

Source	Basis	Equivalent for 60 l test cell
Powrie, Richards et al. (1998)	35 m^3 skip with a recirculation volume of 300-4000 l/day	0.5 to 6 l/day
Reinhart and Al-Yousfi (1996)	300-400 mm/m waste	57 ml/day
Reinhart and Townsend (1998)	1.9m^3/day over 675 m^3 waste at 630 kg/m^3 0.5 m^3/ha/day 5.3 l/m^2/day 1.1 m^3/ha/day	187 ml/day 6 ml/day 633 ml/day 13.2 ml/day
Christensen, Cossu et al. (1989)	3 l/week of water in a modified 44 gallon drum at a density of 25.5-27.7 lb/ft^3	47 ml/day
Šan and Onay (2001)	<2 l per week on a 96 litre sample	<1.6 l/day
Leckie, Pacey et al. (1979)	Minimum of 0.76 m^3/day over 530 ton @1,060 kg/m^3	60 ml/day
Esteban-Altabella, Colomer-Mendoza et al. (2017)	0.25 l over 2.64 kg waste on days 1,7,16,21,23,27,36,45	0.7 l/day

An initial recirculation target of 60 ml/day was selected as a target recirculation volume. Šan and Onay (2001) state that once significant methane levels are reached, and gas is being produced recirculation frequency may be lowered to one or two times per week. Thus it was planned that once gas flow was stablished the volume and timing of recirculation would be reassessed.

Appendix C5 – Compaction

To achieve the densities required in the test cells, small tampers and rammers were used to compress raw waste collected from Kate Valley. Through trial and error in a 151x195 mm square stainless pot and based on the dimension, shape and material of the barrel it was discovered that:

- pneumatic rammers were prone to damaging the side walls of the container if not used carefully to avoid too great a rate. This is due to the heterogeneous nature of the waste material.
- Hand ramming using plates and weights was found to be too cumbersome inside the barrel especially at the bottom where reach was difficult, and the top where space was limited.

- The easiest and quickest method was found to be a 3 kg bar bell or simple block of wood (100 x 50 mm). This ensured a slower rate of compaction requiring on average approximately 36 tamps to achieve the required densities.
- The density of 10 litres of loose raw waste was approximately 832-877 kg/m³
- Compacting 10 litres with a 250 mm long piece of 100x50 mm timber reduced height from 130 mm to 110 mm and a density of approximately 1050 kg/m³
- Compacting 10 litres with a 3 kg barbell (diameter 10 cm) reduced 130 mm to 80 mm and approximately 1300 kg/m³.

Table 29 Verifying compaction methodology

Compactor	Lift height	Weight (g)	Density achieved with 3 kg weight
100x50mm timber	110 mm	3285	1014 kg/m³
100x50mm timber	108 mm	3381	1063 kg/m³
100x50mm timber	109 mm	3306	1030 kg/m³
3 kg bar-bell	87 mm	3333	1302 kg/m³
3 kg bar-bell	87 mm	3357	1311 kg/m³
3 kg bar-bell	88 mm	3402	1312 kg/m³

Using an average density of 1035 and 1308 kg/m³ with the estimated volume of 37 litres gives an estimated weight of waste of 38.3 and 48.4 kg per test cell respectively.

Appendix C6 – Gas Production Rate

Qian, Koerner et al. (2002) supply estimation of gas generation rates for maximum flow rate (which we are interested in) using cumulative calculation based on waste age and mass (p. 355). This can be adapted to the waste used in this experiment as its age and mass of waste is known. Their equation is

$$(Q_i)_t = 2kL_0M_ie^{kt_i}$$

where Q_t = expected gas generation rate for waste mass, M_i, (m³/yr)
k = methane generation constant (yr⁻¹) (range of 0.05-0.15 yr⁻¹)
L_o = methane generation potential (m³/tonne) (range of 140-180 m³/tonne)
M_i = mass of solid waste (tonne)
t_i = age of the waste mass, Mi (yr). Substituting values gives

$$(Q_i)_t = 2 * 0.15 * 180 * 43/1000 * e^{0.15*10}$$

$$(Q_i)_t = 10.4\frac{m^3}{yr} = 28\frac{l}{day} = 0.66\ l/kg\ waste/day\ =\ 0.51\ l/kg\ drywaste/day$$

Appendix C7 – Oxygen Transmission Rates Through HDPE

It is possible to calculate an oxygen transmission rate using fluid transport principles considering absorption of oxygen; diffusion in and through the polymer; and desorption at the opposite side into the test cell. There are numerous complexities in this approach including assumptions of the effect of plasticisers and other additives used, non-Fickian diffusion factors, pressure difference across the HDPE membrane and effect of gas composition expected in the test cell. This indicates a simplified approach may be preferable at this stage.

PolyPrint.com (2021) gives oxygen transmission rates of HDPE at 2300-3100 $cc/m^2/24hr$ at 1mil (1/1000 of an inch) STP which aligns with other sources such as EvalAmericas (2021). However the EvalAmericas (2021) technical bulletin also supplies an oxygen transmission rate of 4448 cc. $25\mu/m^2$.24hr.atm at 35ºC which is more relevant to this experiment. Converting this across the 3 mm thick HDPE test cell assuming a 1 atmosphere pressure difference gives a maximum oxygen transmission rate of 3.2cc/24hr. Based on the approximate 15 litre headspace we would see a rise in oxygen levels of less than 0.02% each day. This is less than the accuracy achievable by the gas analyser. This rate assumes a 1 atmosphere pressure differential across the membrane. This experiment is designed to have positive pressure within the test cell, caused by gas production, creating a pressure differential in the opposite direction. Hence it is expected that the oxygen transmission rate would be much less than this level making the potential effect negligible.

APPENDIX D – Raw Results

Appendix D1 – Test Cell Data

Table 30 Test cell parameters

Test cell	Diameter (mm)	Height (mm)	Volume (litres)	Weight (kg)	Density (kg/m³)
1	390	320	38.2	42.103	1101
2	390	315	37.6	59.806	1589
3	390	315	37.6	41.343	1099
4	390	315	37.6	64.514	1714
5	390	310	37.0	43.316	1170
6	390	315	37.6	57.534	1529
7	390	320	38.2	46.861	1226
8	390	315	37.6	64.638	1718

Appendix D2 – Waste Characteristics

Waste composition range and means based on researcher's data used throughout this thesis. Sources include: Gabr and Valero (1995), Qasim and Chiang (1994), McBean, Rovers et al. (1995), Pohland, Cross et al. (1993), Edjabou (2015), Aguilar-Virgen, Taboada-González et al. (2013), Sel, Çakmakcı et al. (2016), Letcher and Vallero (2019), USEPA (2014), ADEME (1993), Kaza, Yao et al. (2018), Perrot and Subiantoro (2018), NSW EPA (2017), Esteban-Altabella, Colomer-Mendoza et al. (2017).
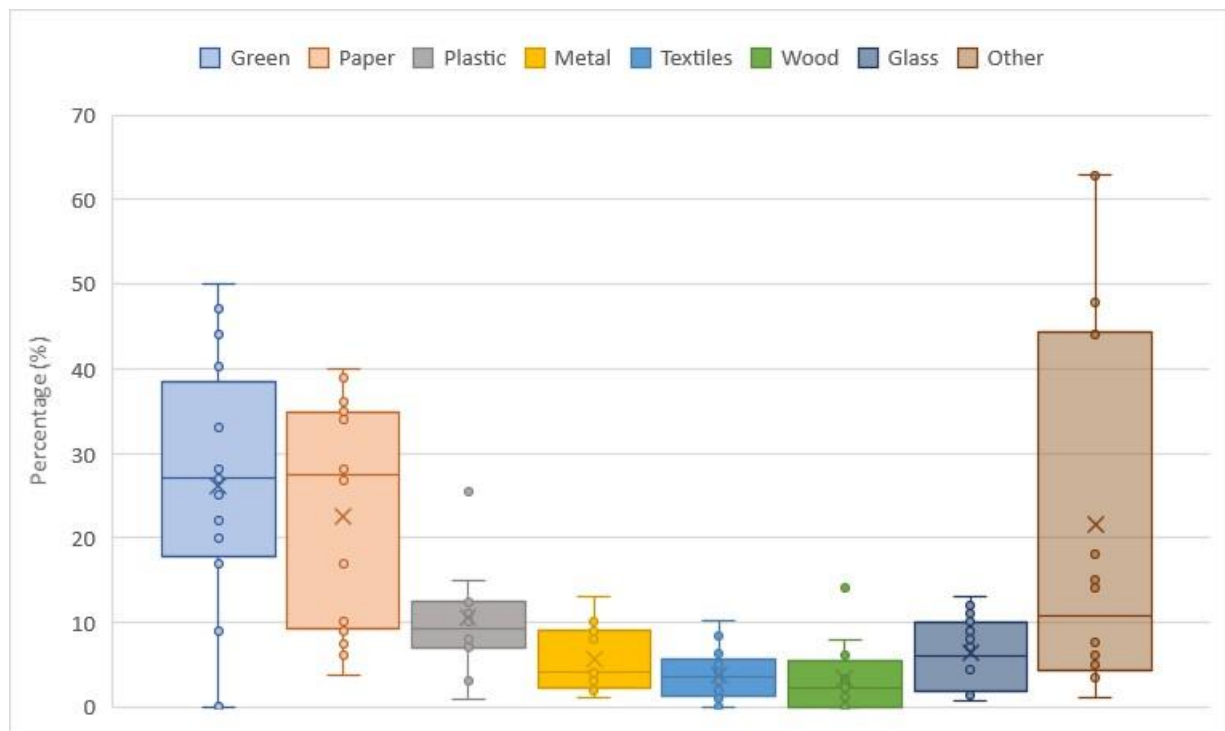
Figure 55 Waste composition raw data in ranges (showing mean as X, median as line, and data points outside the upper and lower quartiles). Note "Other" includes fines, ash, rocks, soil)
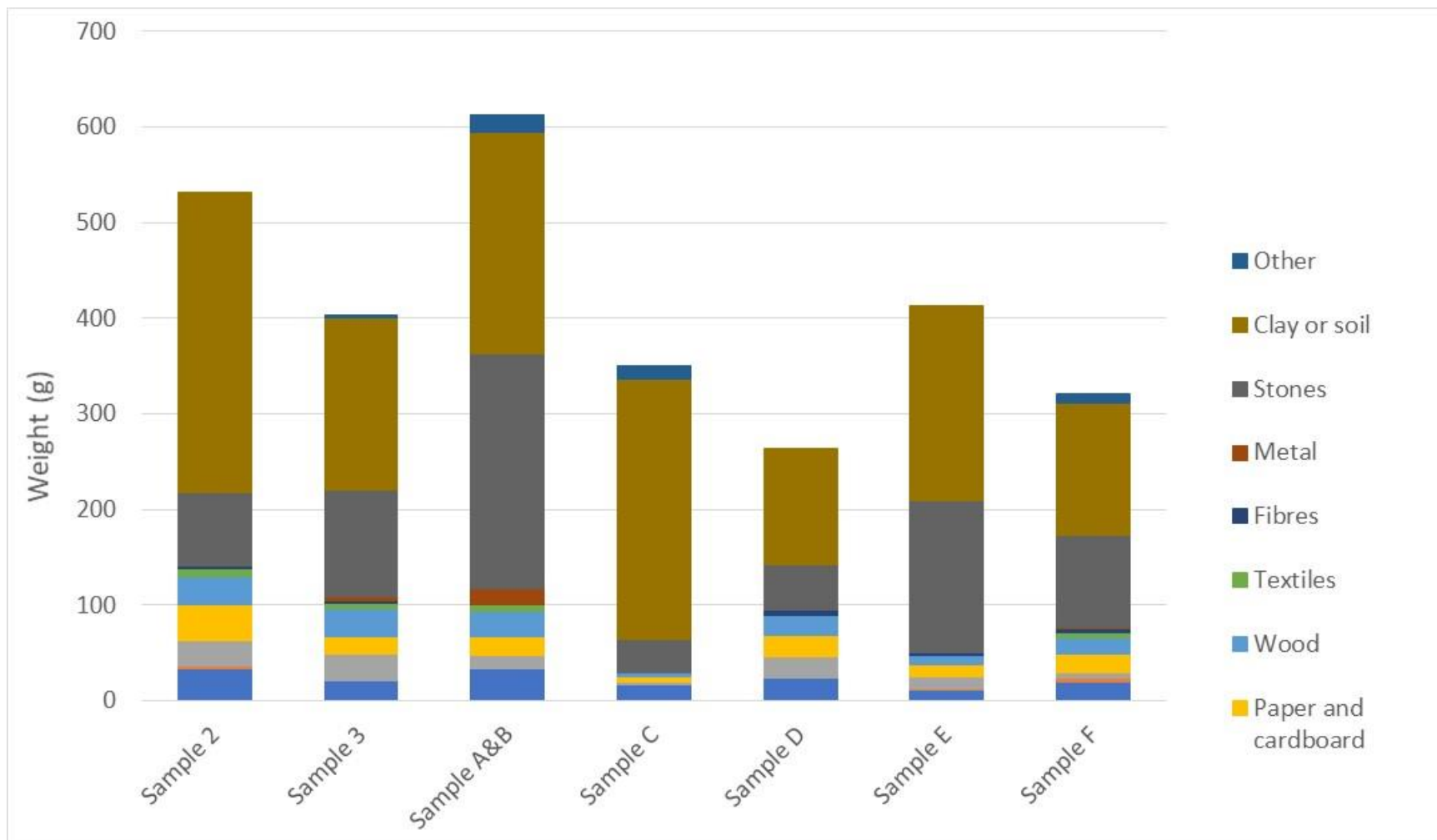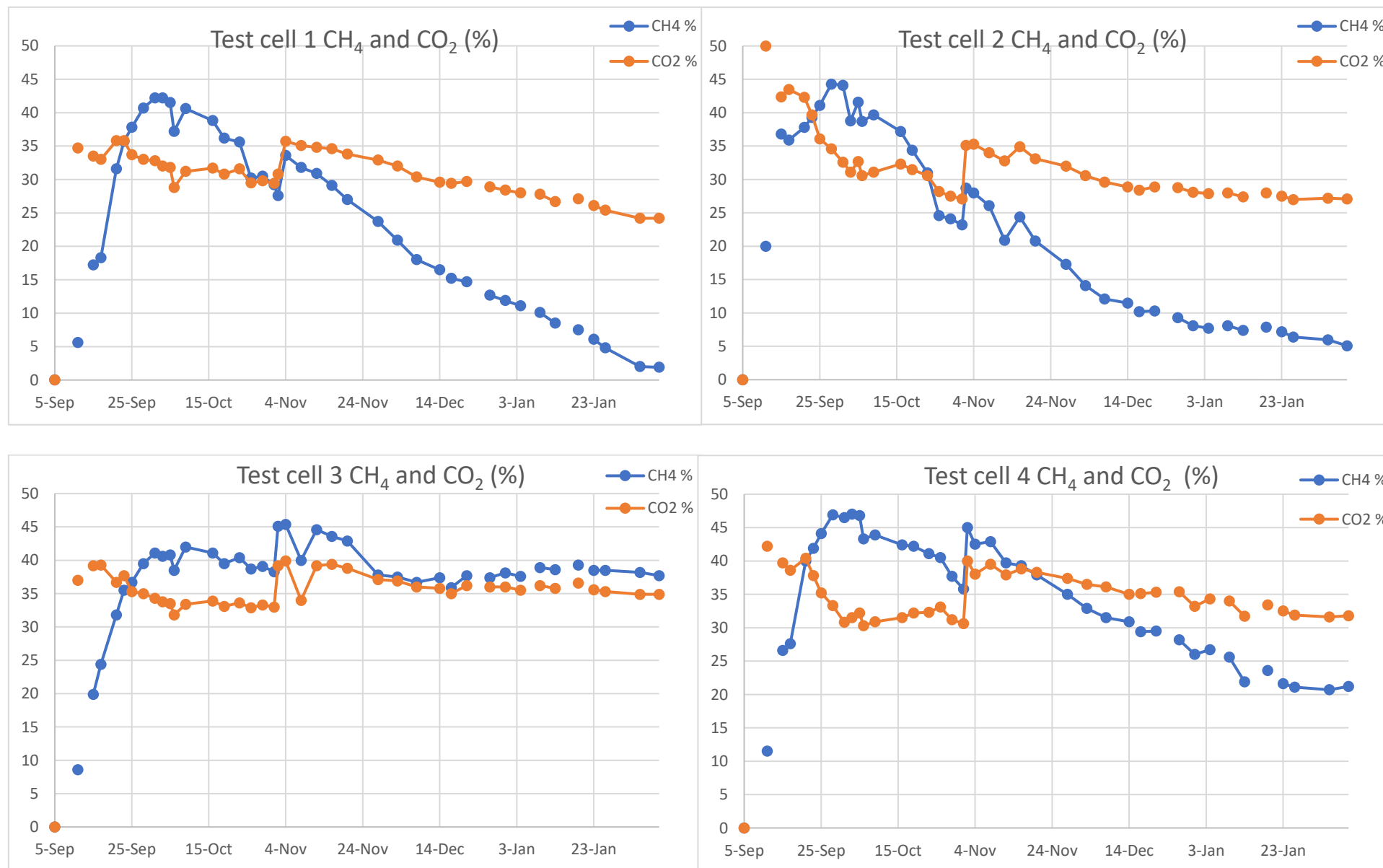
Figure 56 Waste Composition

Appendix D3 – Gas Results

Test cell 1 CH$_4$ and CO$_2$ (%)

Test cell 2 CH$_4$ and CO$_2$ (%)
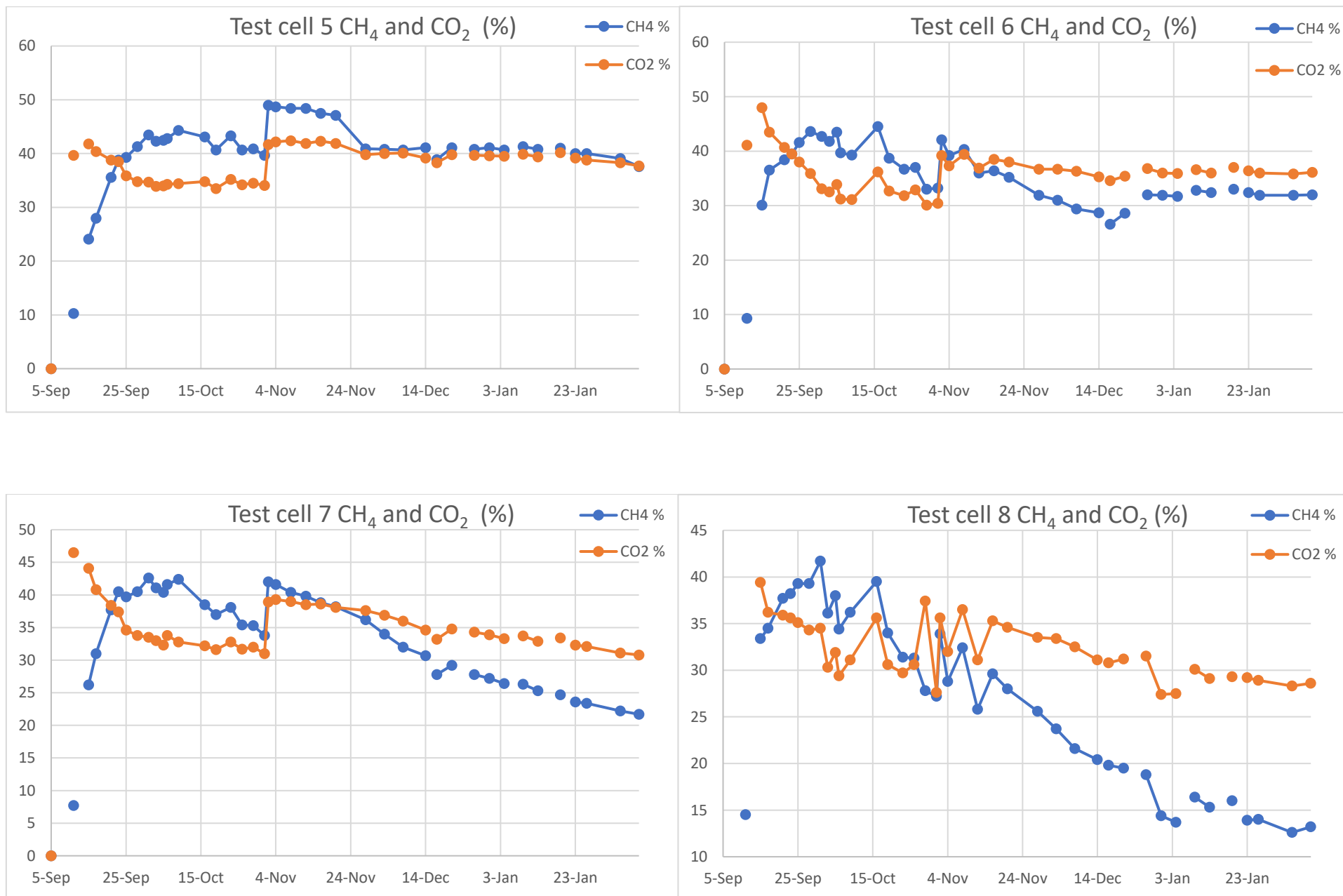
Test cell 3 CH$_4$ and CO$_2$ (%)

Test cell 4 CH$_4$ and CO$_2$ (%)

Figure 57 Methane and CO₂ Gas Composition

Oxygen Levels
Tank 1 and 5 (Low density - intercept leachate)

Oxygen Levels
Tank 2 and 6 (High density - intercept leachate)

Oxygen Levels
Tank 3 and 7 (Low density - std leachate)

Oxygen Levels
Tank 4 and 8 (High density - std leachate)
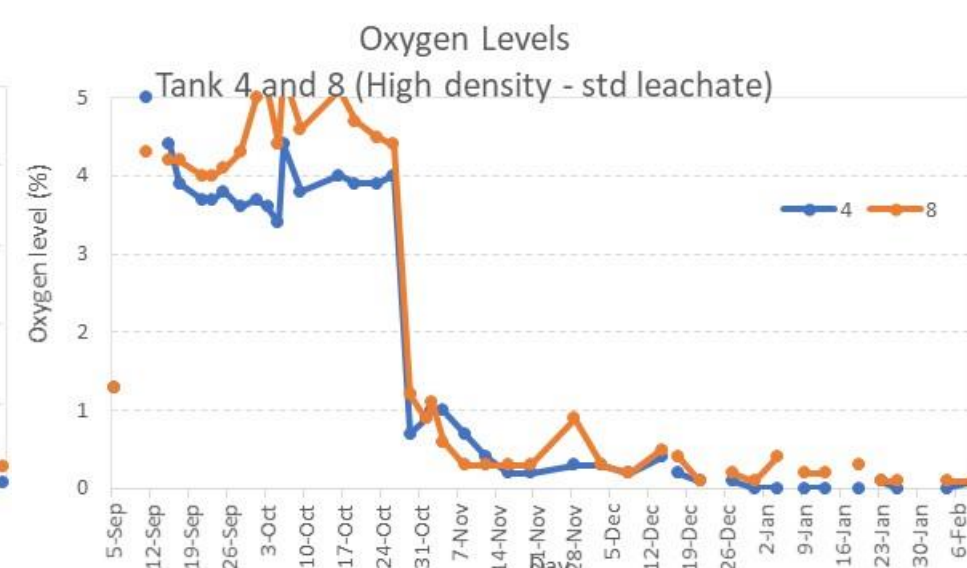
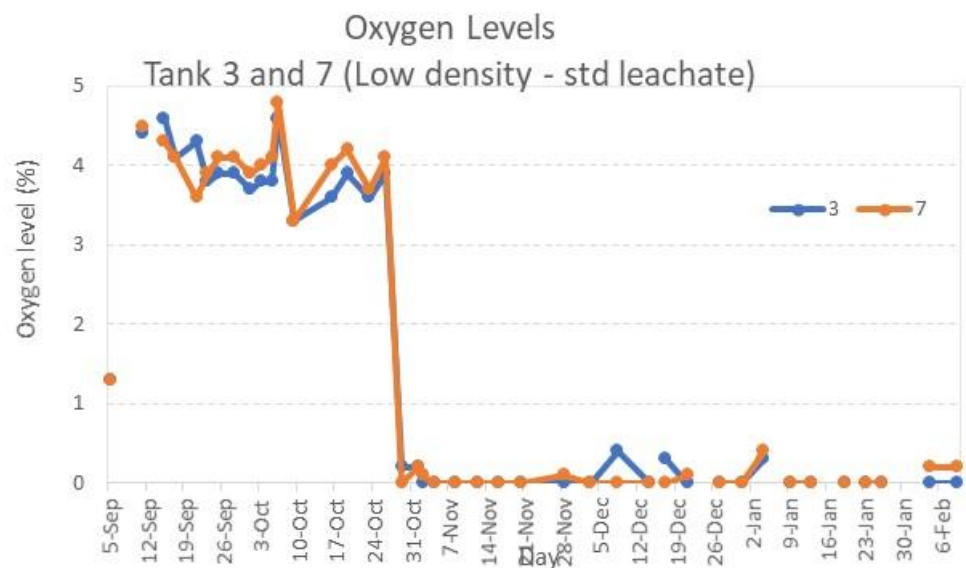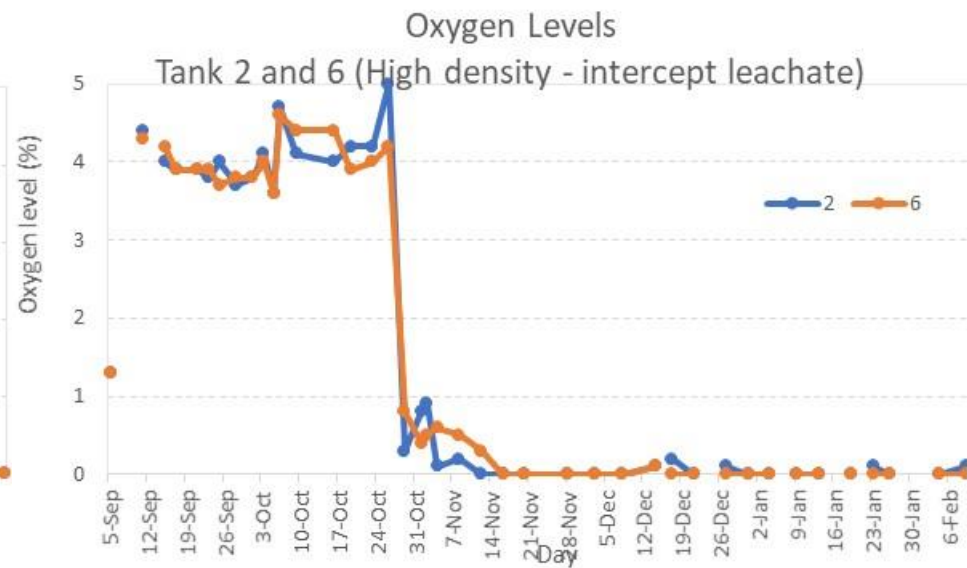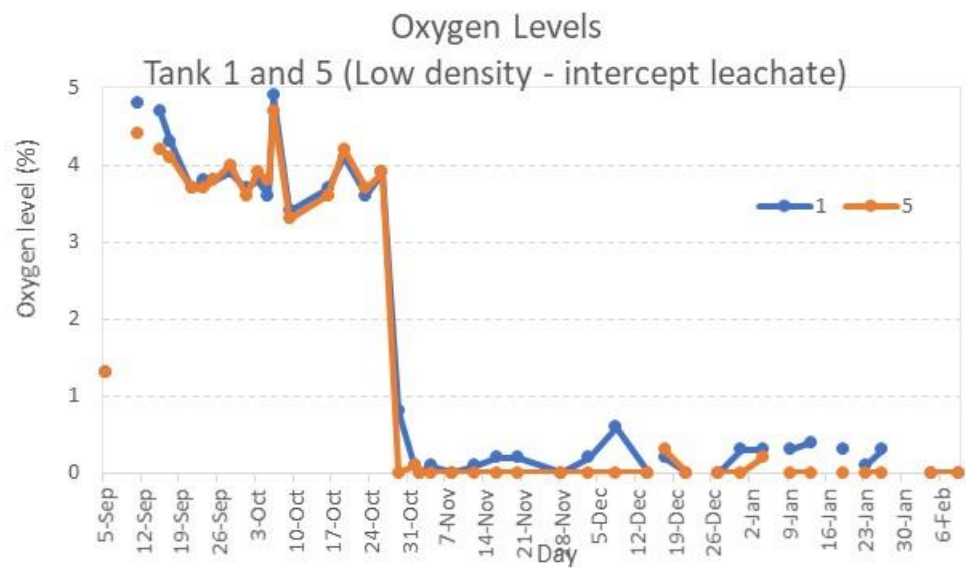## Table 31 Gas analyser error - duplicate data checks

Date	Test cell	Original analyser CH_4 (%)	Second analyser CH_4 (%)	Original analyser CO_2 (%)	Second analyser CO_2 (%)	Original analyser O_2 (%)	Second analyser O_2 (%)
29 Oct	1	30.5	35.6	29.8	35.4	3.9	0.8
29 Oct	2	24.1	30	27.5	35.1	5	0.3
29 Oct	3	39.1	44.7	33.3	37.9	3.8	0.2
29 Oct	4	37.7	47.1	31.2	39.1	4.6	0.7
29 Oct	5	40.9	49.9	34.5	41.7	3.8	0
29 Oct	6	33	41.4	30.1	38.3	5	0.8
29 Oct	7	35.3	43	32	39	4	0
29 Oct	8	27.8	35.4	37.4	35.7	5.7	1.2
1 Nov	1	29.2	35.1	29.4	36.1	4.1	0.1
1 Nov	2	23.2	26	27.1	30	5.2	0.8
1 Nov	3	38.3	45.8	33	39.9	3.9	0.2
1 Nov	4	35.8	45.2	30.6	38.8	5	0.9
1 Nov	5	39.7	49.4	34.1	42.1	4.1	0.1
1 Nov	6	33.2	42	30.4	39.1	4.7	0.4
1 Nov	7	33.8	42.4	31	39.4	4.4	0.2
1 Nov	8	27.2	33.9	27.6	34.6	5.4	0.9

Appendix D4 – Leachate Characteristics

Table 32 Raw leachate results (Hills Laboratory)

Hill Laboratories
TRIED, TESTED AND TRUSTED

R J Hill Laboratories Limited
28 Duke Street Frankton 3204
Private Bag 3205
Hamilton 3240 New Zealand

T 0508 HILL LAB (44 555 22)
T +64 7 858 2000
E mail@hill-labs.co.nz
W www.hill-laboratories.com

Certificate of Analysis
Page 1 of 3

Client:	University of Canterbury	Lab No:	2492208	SPv1
Contact:	Kees Beentjes	Date Received:	11-Dec-2020	
	C/- University of Canterbury	Date Reported:	29-Dec-2020	
	Private Bag 4800	Quote No:	108927	
	Christchurch 8140	Order No:	691655	
		Client Reference:	Landfill leachate	
		Submitted By:	Kees Beentjes	

Sample Type: Aqueous

Sample Name:		Raw Leachate 06-Dec-2020 12:15 pm				
Lab Number:		2492208.1				
Individual Tests						
Total Suspended Solids	g/m³	31	-	-	-	-
Total Dissolved Solids (TDS)	g/m³	3,700	-	-	-	-
Settleable Solids	mL/L	< 1.0	-	-	-	-
Total Calcium	g/m³	45	-	-	-	-
Dissolved Iron	g/m³	0.9	-	-	-	-
Total Iron	g/m³	1.07	-	-	-	-
Total Magnesium	g/m³	30	-	-	-	-
Total Potassium	g/m³	350	-	-	-	-
Total Sodium	g/m³	800	-	-	-	-
Chloride	g/m³	970	-	-	-	-
Total Ammoniacal-N	g/m³	920	-	-	-	-
Nitrite-N	g/m³	< 0.10 #2	-	-	-	-
Nitrate-N	g/m³	< 0.10	-	-	-	-
Nitrate-N + Nitrite-N	g/m³	< 0.10 #2	-	-	-	-
Dissolved Reactive Phosphorus	g/m³	8.1	-	-	-	-
Phosphate	g/m³	25	-	-	-	-
Sulphate	g/m³	7 #5	-	-	-	-
Total Biochemical Oxygen Demand (TBOD₅)	g O₂/m³	25	-	-	-	-
Chemical Oxygen Demand (COD)	g O₂/m³	1,720	-	-	-	-
Dissolved Organic Carbon (DOC)	g/m³	640	-	-	-	-
Total Organic Carbon (TOC)	g/m³	670	-	-	-	-
Volatile Fatty Acid Profile						
Volatile Fatty Acids (VFA), Total	g/m³ as acetic acid	< 10	-	-	-	-
Formic Acid	g/m³	< 5 #1	-	-	-	-
Acetic Acid	g/m³	< 5 #1	-	-	-	-
Propionic Acid	g/m³	< 5 #1	-	-	-	-
Butyric Acid	g/m³	< 5 #1	-	-	-	-

Table 33 Heavy metal analysis of raw leachate

Analyte	Sample 1 Concentration (µg/L)	Sample 2 Concentration (µg/L)
Na	11,212.565	6,451.91
Mg	411.345	257.754
Al	15.868	11.424
P	140.111	90.011
K	4,560.056	2,750.002
Ca	715.629	470.069
Cr	10.579	6.382
Mn	3.578	2.247
Fe	33.724	23.86
Co	0.318	0.177
Ni	16.205	15.821
Cu	3.035	2.044
Zn	5.428	5.229
As	8.013	4.717
Cd	0.134	0.169
Pb	0.208	0.02

Table 34 Attempt at colour measurement on Hach analyser

Test cell	Colour reading
1	over-range
2	364 nm
3	over-range
4	over-range
5	over-range
6	over-range
7	over-range
8	359 nm
Raw leachate	over-range

Table 35 Volatile solids sample results

Sample	% volatile solids (dry weight)
Raw1	9.8%
Raw1	9.5%
Raw1	8.8%
Raw1	8.7%
Raw 3	10.1%
Raw 3	7.5%
Raw 3	10.2%
Raw 3	8.8%
Raw 2	12.2%
Raw 2	11.3%
Raw 3	9.1%
Raw 3	7.1%

Appendix D5 – Biomethanation Potential Setup

Table 36 Test cell setup analysis

Sample	Moisture content	Volatile Solids % dry
Test cell 3a	23%	6.8%
Test cell 3b	27%	10.2%
Test cell 3c	22%	7.8%
Test cell 5a	25%	8.3%
Test cell 5b	26%	10.0%
Test cell 5c	26%	9.8%
Test cell 4a	24%	9.4%
Test cell 4b		8.1%
Average	**25%**	**8.8%**

Table 37 Tube substrate weights

Tube Number	Tube Sample	Weight of waste (g)	Tube Number	Tube Sample	Weight of waste (g)
1	DSS only	0	14	DSS only	0
2	DSS only	0	15	DSS only	0
3	Rice	20.1	16	Rice	20
4	Rice	20.0	17	Rice	20
5	Raw Waste	885.7	18	Refresh Waste	889.9
6	Raw Waste	885.7	19	Refresh Waste	884.5
7	Raw Waste	885.4	20	Refresh Waste	884.2
8	Test Cell #5	887.9	21	Refresh +Rice	885+20
9	Test Cell #5	888.9	22	Refresh +Rice	885.9+20
10	Test Cell #5	889.7	23	Raw Waste	885.6
11	Test Cell #3	886.3	24	Raw Waste	885.8
12	Test Cell #3	886.7	25	Raw +Rice	884.5+20
13	Test Cell #3	888.3	26	Raw +Rice	884.7+20

- Includes all caps and valves with 10 mm fittings and 6 mm-10 mm adapter

Appendix D6 – Biomethanation Potential Results

Note scales vary to allow the raw data to be examined.
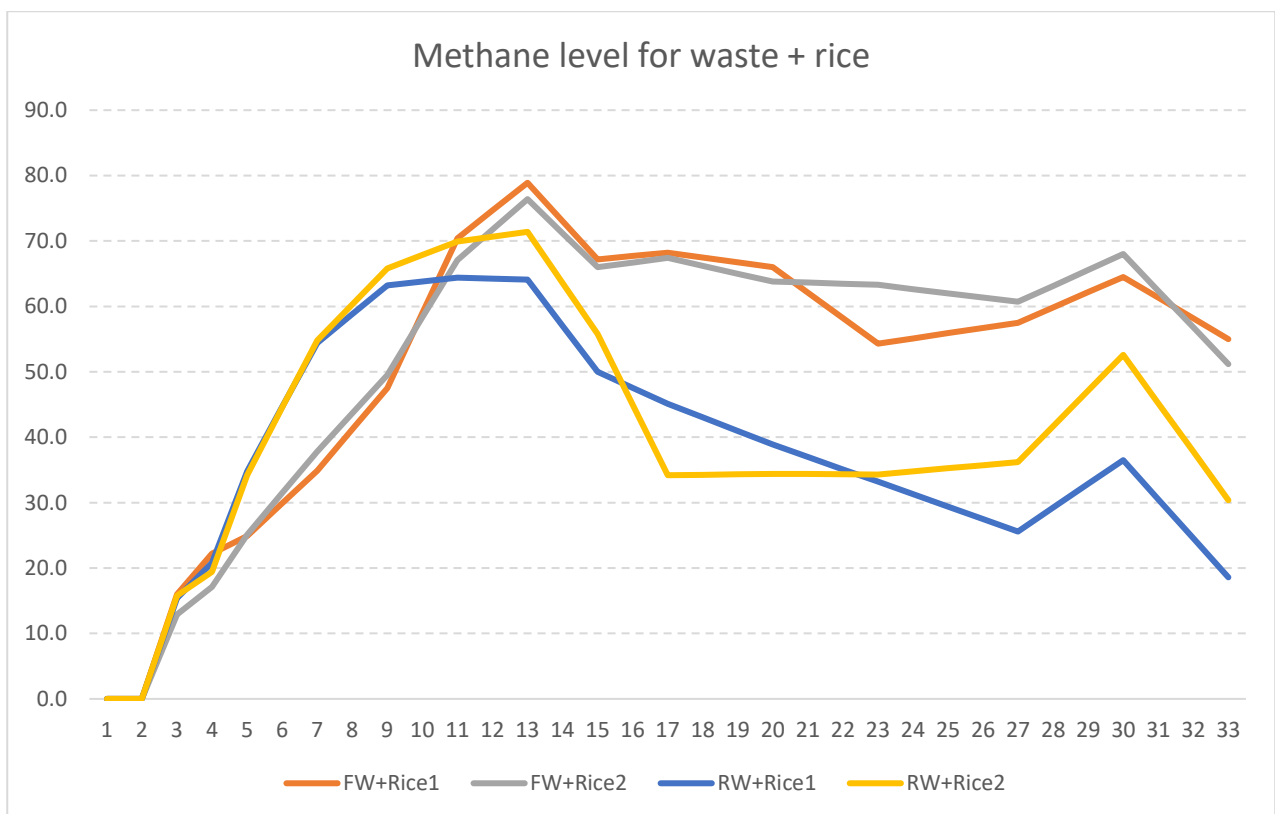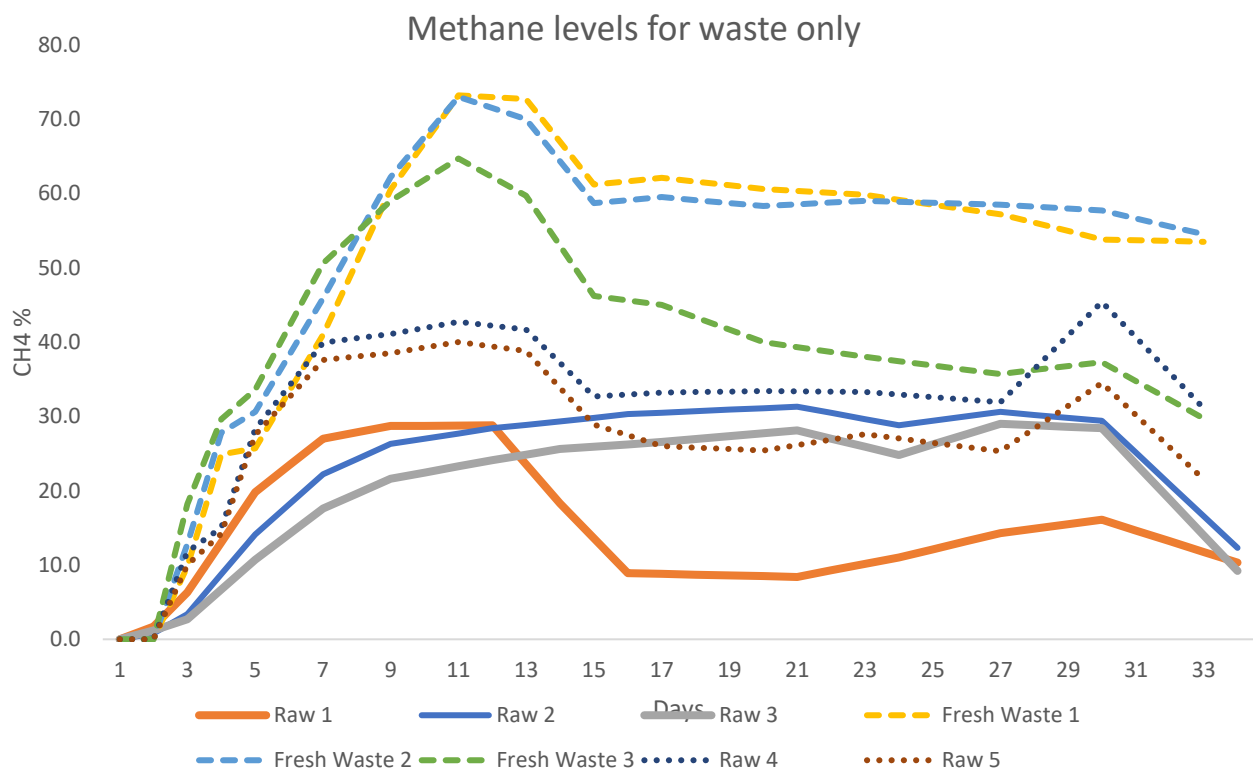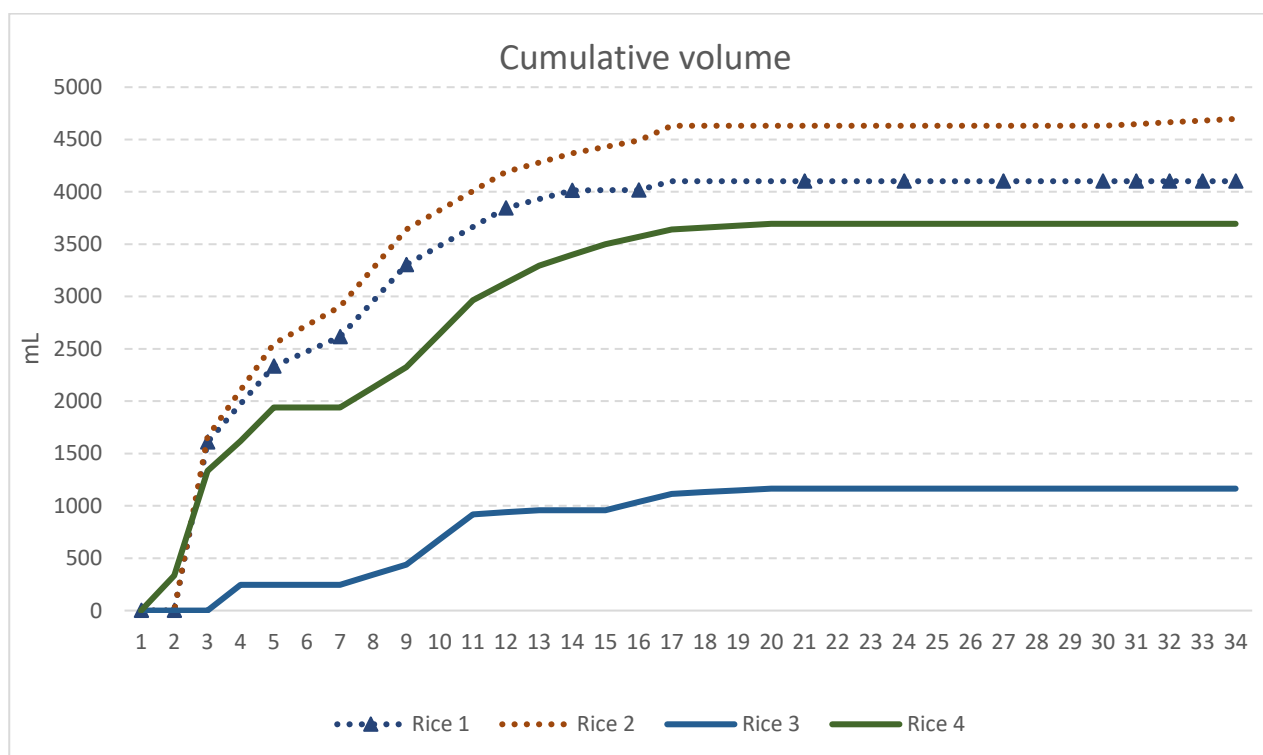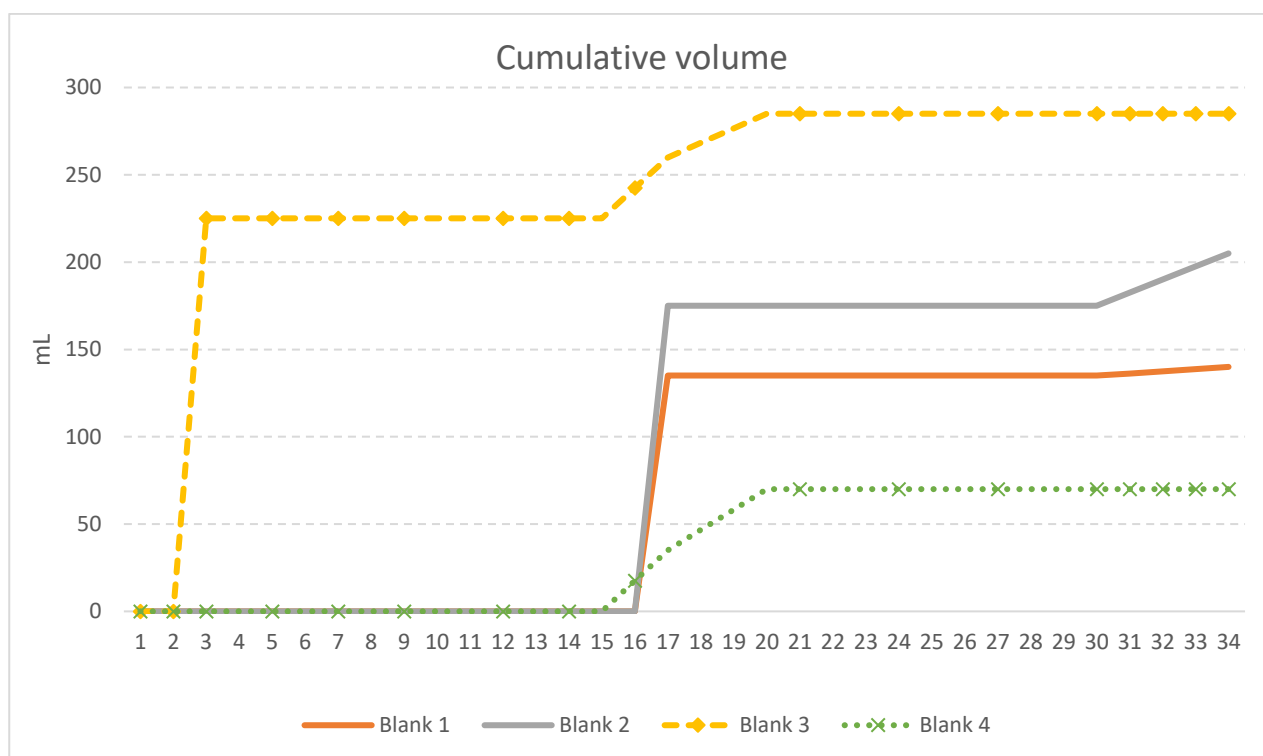

Methane levels for blanks only

Methane levels for rice only
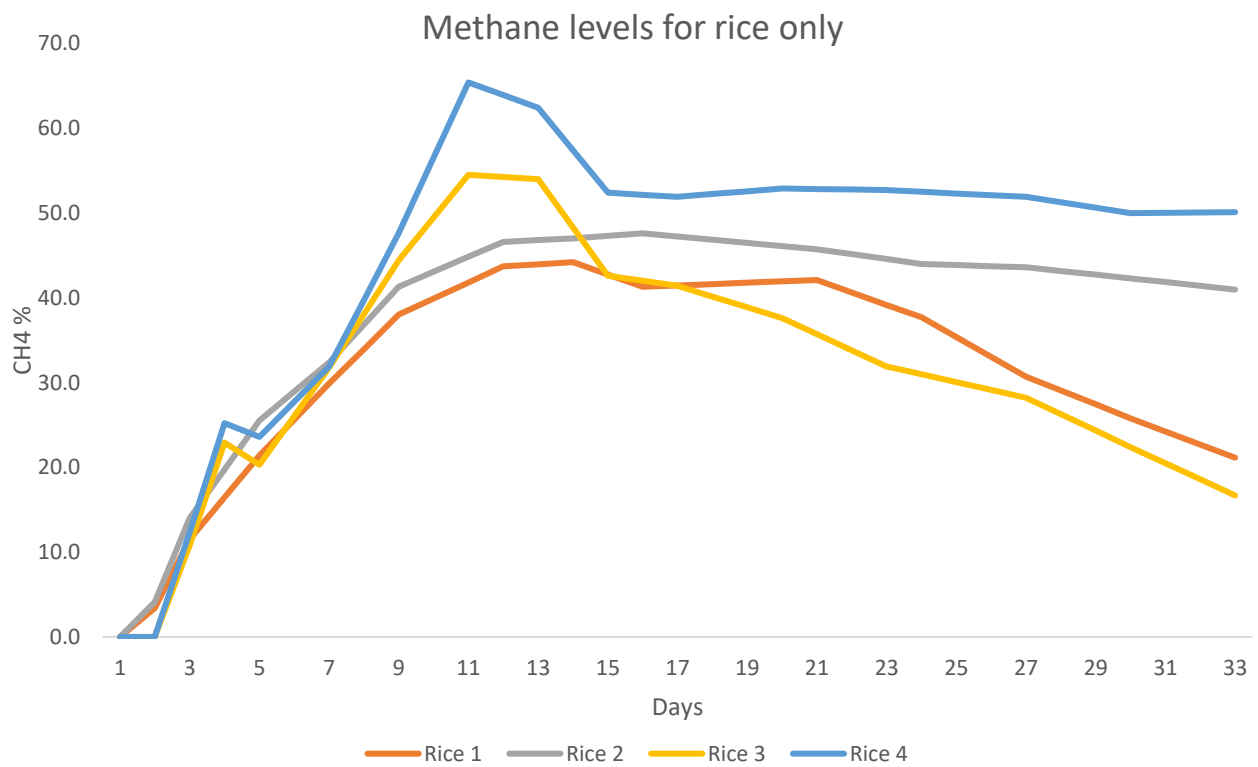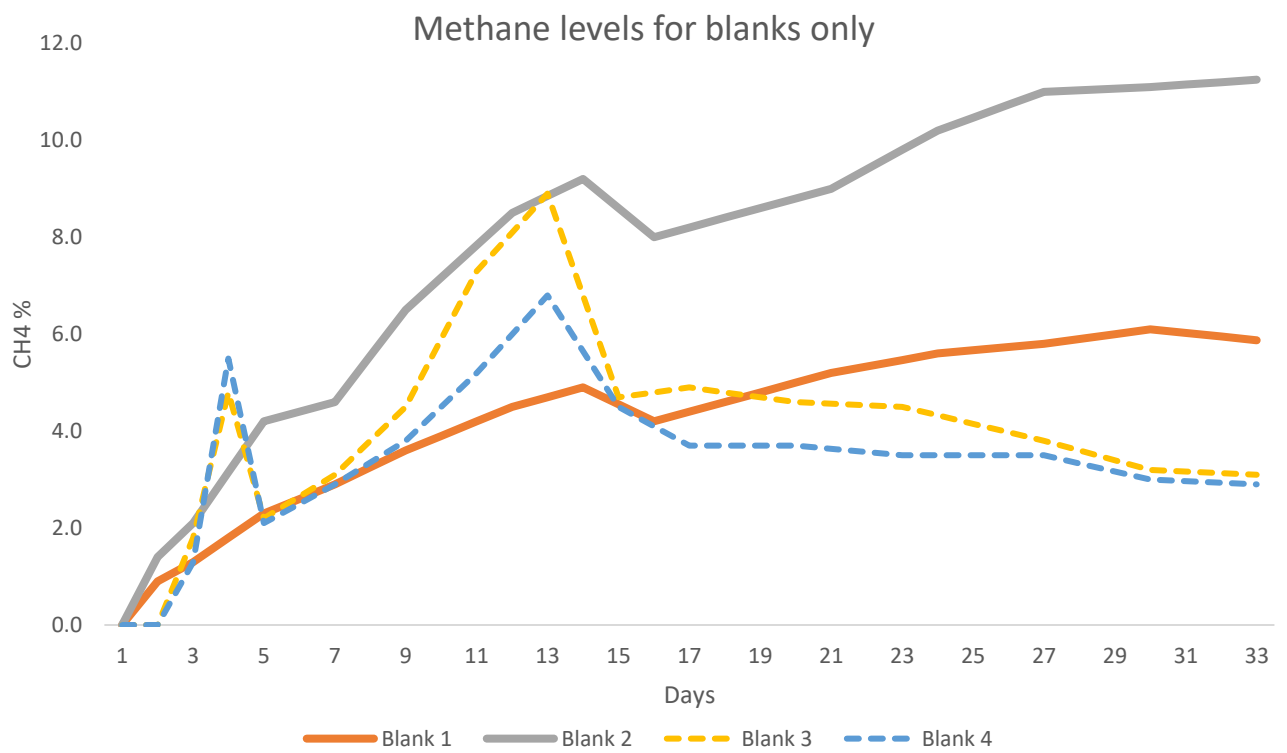
Figure 58 Methane levels recorded

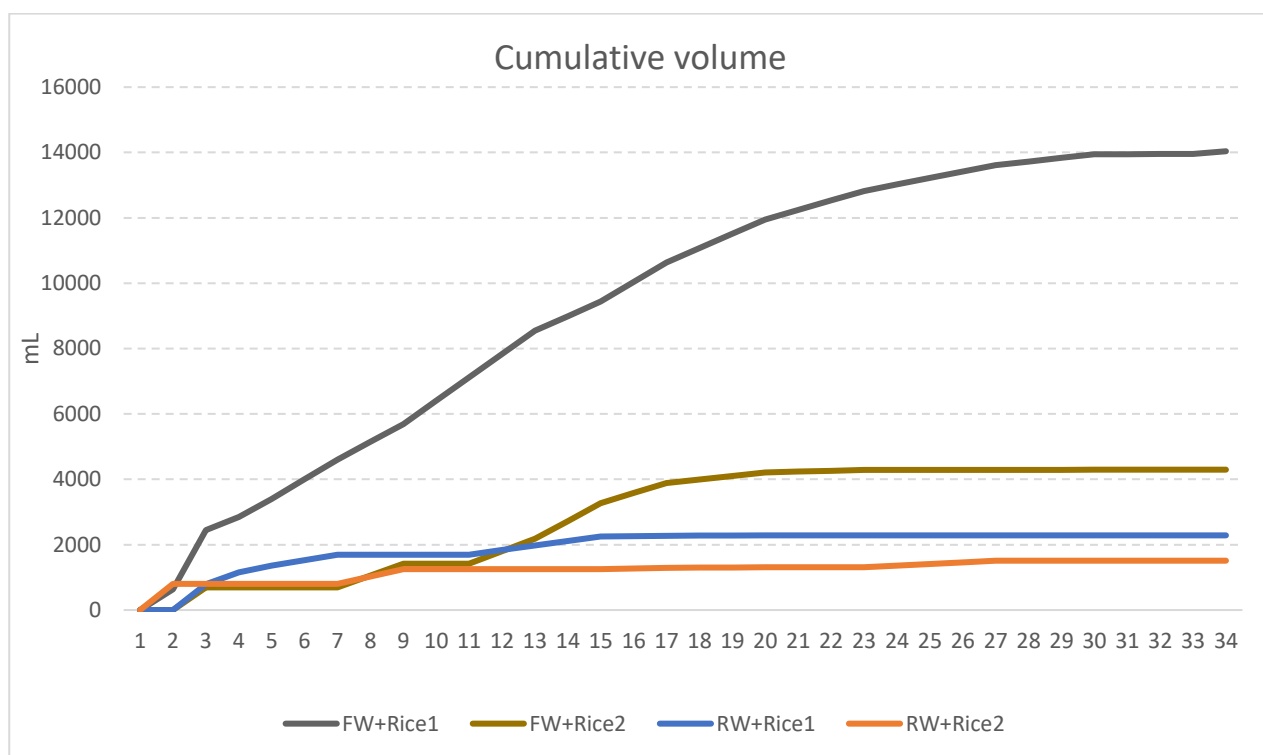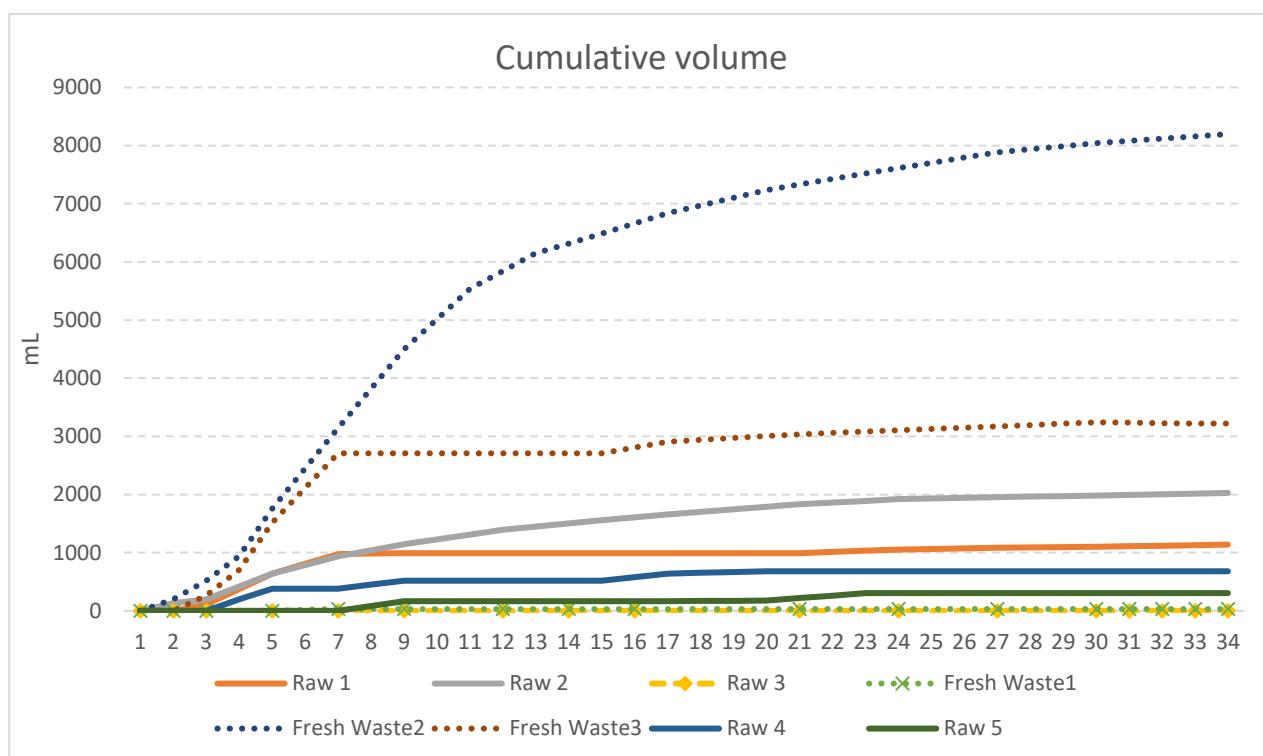Cumulative volume

Cumulative volume

Figure 59 Gas volumes recorded

APPENDIX E – Mechanisms of Chemical Movement in Leachate

Daniel (1993) and Qasim and Chiang (1994) outline processes of attenuation and migration of chemical constituents with respect to groundwater and soils. The same processes are expected to apply in landfills to some extent. These processes include:

- Diffusion
- Dispersion
- Dilution
- Filtering effects
- Physical and chemical sorption

Diffusion is caused by the movement of the chemical compound from an area of high concentration to that of lower concentration. It can be a significant mechanism if leachate flow rates are low, especially in localised areas.

Dispersion is associated with leachate velocity within the pores of the waste matrix. It is not an effective mechanism when considering bulk leachate properties (Lu, Eichenberger et al., 1984).

Dilution is a function of the water content of the waste which may change temporally and spatially.

Filtering effects covers the removal of suspended solids by filtering mechanisms of the waste. This may include attenuation of non-polar organic compounds (oil or other hydrocarbons) and other mechanisms such as mechanical entrapment, gravity settlement, and absorption.

Physical sorption is a function of the forces occurring in soil or waste particles. Van der Waals force, hydrodynamic and electrokinetic properties influence physical sorption which affects bacteria and viruses more than chemical contaminants. However Lu, Eichenberger et al. (1984) maintain **chemical sorption** is more effective with chemical compounds and summarise five chemical mechanisms in landfills:

- Precipitation and dissolution
- Adsorption/desorption
- Complexation/Mineralisation
- Ion exchange
- Redox

Precipitation and dissolution occurs when chemical reactions are not in equilibrium within the leachate. This means particles are solubilising or precipitating into or from the waste matrix. This is an important mechanism for metal compounds and is particularly relevant with calcium and magnesium in clogging mechanisms within the landfill.

Adsorption/desorption is the most common mechanism with trace contaminants, and like precipitation and dissolution is affected by pH and redox conditions. Adsorption rates are often much faster than precipitation/dissolution and affect clay minerals in particular.

Complexation involves the reaction of metal ions with inorganic anions and organic ligands to form complex inorganic ions or organometallic complexes. Complexes can occur in the solution phase of the leachate increasing the level of complexes in the leachate or can develop on solid surfaces effectively lowering those levels in the leachate. The chemistry involved is multifarious and concentrations are not the result of equilibrium processes. Therefore, the attenuation or migration of complex species in leachate can be difficult to evaluate with any degree of certainty. **Mineralisation** is the conversion of organic material into inorganic states. In a similar manner through biological assimilation, inorganics may be converted (and immobilised) into microbial tissue.

Most **ion exchange** effects originate from exchange sites on organic matter or silicate clays, usually occurring on the surface of clay minerals and colloidal organics. It is typically an equilibrium exchange to maintain neutrality however the nature of these types of reactions means they can re-exchange with different species depending on affinity and charge. This re-exchange particularly affects trace metals as landfill leachates are dominated by calcium, magnesium, sodium and potassium ions. Cation Exchange Capacity is affected by the minerals in the clay, type of organic molecules, pH and available surface area. The porosity of the waste and the flow rate of the leaching fluid are factors in regulating the migration of ions.

Redox reactions occur when the redox potential of the leachate is different to that of the solutions at or near the waste particles. During the movement of leachate through landfills, changes in redox potential change the oxidation state and chemical forms of many chemical species in both solution and solid phases of the system. These reactions affect the solubility of complexes and initiate or disturb precipitation/dissolution mechanisms.

Microbiological reactions which cause biodegradation affect redox, as they affect the other mechanisms of mineralisation, immobilisation, precipitation/dissolution and complexation. The redox potential of leachates from landfills undergoing biodegradation are usually reducing, however redox potential changes with mode of degradation (methanogenesis, acid-forming, aerobic) and is known to differ with organic species. Control of and overcoming energy barriers associated with redox reactions is developing into an important field in waste treatment (Dang, Holmes et al., 2016). Technology like electromethanogenesis is attracting considerable research to overcome scale-up issues and is represented as one of the most promising applications of bioelectrochemical systems. Direct interspecies electron transfer using conductive materials such as carbon nanotubes, biochar, granular activated carbon, and magnetite have been showed to be highly efficient in enhancement of methane yield (Gahlot, Ahmed et al., 2020).

APPENDIX F – Microbial Diversity

Table 38 outlines the key physiological groups of microbes involved in anaerobic degradation.

Table 38 Key microbes involved in anaerobic degradation
Source: Christensen, Cossu et al. (1989)

Microbial group	Substrate used
Amylolytic bacteria	Starches
Proteolytic bacteria	Proteins
Cellulolytic bacteria	Cellulose
Hemicellulolytic bacteria	Hemicellulose
Hydrogen-oxidising methanogenic bacteria	Hydrogen
Acetoclastic methanogenic bacteria	Acetic Acid
Sulphate-reducing bacteria	Sulphate

Sekhohola-Dlamini and Memory (2020) introduce the microbiology of MSW with numerous references but state:

> due to the heterogeneous nature of municipal waste, landfills are reputedly one of the most dynamic terrestrial ecosystems harbouring highly diverse populations of microorganisms with capacity to coexist and metabolise complex substrates.

The microbial community responsible for degradation in all phases are multiple, complex, variable, cooperative and are comprised of different taxonomic kingdoms and different phyla. Bacteria, archaea, and fungi are typically sequenced during studies as they are known to participate in aerobic and anaerobic degradation. However, viruses, protozoa, oomycetes and even algae are known to exist in landfills although their participation in degradation, along with other fungi, has not been clearly established.

Microbial diversity in municipal landfills is predominantly identified at phylum level, with few studies reporting species composition. Bacterial communities are by far the most profiled microorganisms in municipal landfills; with members of bacterial phyla Proteobacteria, Firmicutes and Bacteroidetes being the most proliferate and dominate populations identified in landfills. Sekhohola-Dlamini and Memory (2020) however include in their review that preferences in profiling bacteria and weaknesses in techniques of identifying other distinct phyla may be skewing the notion that bacteria dominate landfill biota.