SUSTAINABLE WASTE MANAGEMENT THROUGH OPERATING LANDFILL AS BIOCELL

by

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Abstract

SUSTAINABLE WASTE MANAGEMENT THROUGH OPERATING LANDFILL AS BIOCELL

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Solid waste is being generated at a record pace, and disposal of it requires advanced waste management services at a reasonable cost. Landfilling is the most suitable waste disposal and management technique currently available, and it is being used throughout the world. It is, however, the largest anthropogenic source of atmospheric methane, and it requires a lot of space.

The primary objective of this study was to investigate an alternative sustainable solution to waste management by operating landfills as biocells. Biocells are cost effective, produce methane more rapidly, and accelerate space recovery. A laboratory scale study was conducted to investigate the effects of enzymes and manure on solid waste decomposition and gas production in a landfill biocell. To simulate the biocell, laboratory scale reactors were filled with municipal solid waste (MSW) and food waste, and the manganese peroxidase (MnP) enzyme, three types of manure (from cows, pigs and horses), and sludge were used as inoculum. MSW reactors with MnP produced the highest amount of methane, followed by reactors with pig manure. Among the food waste reactors, the highest methane volume was generated by reactors with cow manure. But all of the food waste reactors produced less methane than the MSW reactors, due to the long lag phase. Based on the results from the laboratory scale study, two field scale test

cells (control cell and biocell) were installed in the City of Denton Landfill with MSW feedstock and were monitored for almost 14 months. Though reactors with MnP produced the highest amount of methane in the laboratory, it was not used in field. Instead, a combination similar to that of the MSW reactor with pig manure was used in the field biocell, where the control was simulated as a bioreactor landfill. The results from the field experiment revealed that the biocell test section that was fed with organic fractions of MSW, pig manure, and sludge produced nearly three times more the amount of methane (12,437 standard cubic feet) than the control section (4,644 standard cubic feet). The estimated decay rate of the biocell was considerably larger (1.32 $year^{-1}$) than the decay rate of control cell (0.18 $year^{-1}$) and the other values found in literature (0.003 to 0.21 $year^{-1}$). The quality and amount of the landfill biogas and quality of the leachate showed that the pig manure enhanced the MSW biodegradation in both laboratory scale landfill simulation and field application. Thus, it can be concluded that operating a landfill as a biocell is a sustainable waste management system that results in enhanced methane production and waste decomposition.

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Chapter 1

Introduction

1.1 Background

Solid waste generation is increasing alarmingly with the rapid growth of urbanization, and it is causing a huge increase in demand for waste management services (Bhuiyan, 2010). Solid waste management directly influences public health, safety, and the environment, as its improper management poses serious threats to natural resources and retards efficient sustainable development (Kumar and Bhowmick, 1998). At present, the world's cities generate about 2 billion tons of municipal solid waste (MSW) per year, and that amount is expected to double by 2030 (GWMO-ISWA, 2015). Approximately 254 million tons of wastes were generated in 2013 in the USA, averaging 4.40 pounds per person per day, of which 52.8% was discarded in landfills (US EPA, 2015).

Organic waste comprises the largest share of MSW in low-income countries (50% to 70%); in high-income countries, it is typically 20% to 40% (GWMO-ISWA, 2015). Organic materials are the largest component of MSW in the USA, where food waste averages 14.6%, paper and paperboard account for 27%, and yard trimmings contribute about 13.5% (US EPA, 2015). In East Asia and the Pacific region, approximately 270 million tons of solid waste are generated per year (Hoornweg et al., 2013), mainly composed of organic waste (Shekdar, 2009). Organic waste, especially food waste, is responsible for a major share of soil and water pollution, as well as greenhouse gas (GHG) emissions, which are estimated to be 4.14 tons of carbon dioxide equivalent GHG from per ton of food wasted (Oelofse and Nahman, 2013) and are a matter of serious environmental concern. It is also a great source of biogas, as it has the potential for generating large amounts of methane, which is harmful for environment since it has 21 times greater global warming potential than carbon dioxide (IPCC, 1996). Organic fractions of municipal solid waste (MSW) contribute approximately 0.2-0.6 billion tons of greenhouse gasses (GHG) to the atmosphere every year (Manfredi et al., 2009, World Bank, 2012). According to Tahir et al. (2015), solid waste generation and management is a burning issue all over the world, contributing 3% of the total GHG globally.

Although the recycling rate has increased in recent years, landfilling is still the most dominant waste management practice in the USA (USEPA, 2015) and worldwide, as it is the simplest, cheapest, and most cost-effective waste disposal method available (Barrett and Lawler, 1995). Landfills are the largest anthropogenic source of atmospheric methane in many countries. For example, 30% and 24% of landfill gasses are emitted from landfills in Europe and the United States, respectively (Nikiema et al., 2007). The reduction of greenhouse gas (GHG) emissions to the atmosphere is one of the key challenges around the world, and the conversion of methane to energy might be the best possible solution. A sustainable waste management system requires reduced energy and resource consumption, as well as increased recycling and reuse of materials, to decrease the amount of waste dumped in landfills (MoE Japan, 2006). The significant volume of organic waste in the waste stream and its potential for generating large amounts of methane mandates diverting waste from landfills.

An anaerobic digester (AD) is an alternative method for organic waste treatment, but it is expensive and requires technical expertise to operate. The high solid content, large particle size and heterogeneous nature of MSW also makes process control difficult in an AD (GWMO-ISWA, 2015). A biocell is another option, and it is cost effective and sustainable. It is predicted that the cost of solid waste management will increase from today's annual $205.4 billion to about $375.5 billion by 2025 globally (World Bank, 2012). It is, therefore, vital to find a sustainable, cost effective way to manage our MSW.

1.2 Problem Statement

Sanitary landfilling is an engineering method for disposing of solid waste in a safe manner. The oldest practice of waste disposal, open dumps, impacted the environment negatively was not sustainable in terms of resource utilization (Hettiaratchi et al., 2007). Dry tomb landfills were the next step towards the land disposal of solid waste, in which the waste was kept dry to minimize leachate production. In contrast to both open dumps and dry tomb landfills, landfill bioreactors use specific design and operational practices to enhance waste biodegradation and the gas production rate, while minimizing environmental impacts (Yuen, 2001; Reinhart et al., 2002). Enhancing the rate of waste biodegradation in landfills provides a number of benefits, such as producing more gasses that can be used as an energy source, cost effective leachate management, and the opportunity to reuse the landfill space (Pacey et al., 1999). Landfill bioreactors can be operated anaerobically or aerobically, or in hybrid mode (sequential or simultaneous anaerobic/aerobic modes). Sequential operation has the advantage of energy, resource, and space recovery if the stabilized waste is mined at the final stage (Hettiaratchi et al., 2007). Although bioreactors have been proven successful over the past few decades, there is still a need for new landfill facility once the active period of the bioreactor landfill is over. For over-populated regions like Asia, Africa, and Latin America, and even for Europe, landfill space is an emerging issue. In North America, regulations require post-closure activities and financial assurance for 30 years after landfill closure, and a state agency may require additional years of care, if needed (US EPA, 2001).

To address the problems related to space, a variation of the bioreactor, the "sustainable biocell" or "biocell" was proposed by Hettiaratchi (2007). The biocell is frequently referred to as the third generation of landfills. It differs from traditional landfills by operating as a temporary facility, rather than a permanent one (Bartholameuz, 2015), and the space is reused (Hettiaratchi, 2007), thus eliminating the need for post-closure monitoring. According to Hettiaratchi et al. (2010), a sustainable landfill biocell simultaneously solves the problems associated with slow degradation of MSW and space recovery. The biocell involves sequential operation of a landfill cell, producing methane gas during the first stage of anaerobic degradation, in-situ composting within the cell footprint during aerobic degradation in the second stage, and landfill mining for resources and space recovery in the third stage (Meegoda, 2013). Unrecovered waste has a high energy content that can be used as refuse-derived fuel, such as an energy pallet. Some lab-based studies have shown the effect of augmenting leachate by adding enzymes (Jayasinghe et al, 2011), sludge (Warith, 2001; Alkaabi et al., 2009) and horse manure (Yazdani, 2010). etc. to achieve enhanced biodegradation of waste. Manures can be effective additives that promote faster decomposition during the early stages of landfilling (Yazdani, 2010); enzymes can be used to break down the lignin content in waste during the later stages (Jayasinghe, 2011).

Leachate augmentation, using enzymes and sludge, was performed in lab scale studies, but was not feasible in the field. In addition, enzymes are very expensive, so they may not be economically feasible for developing countries where manure is a viable option. This study focuses on investigating how adding enzymes and manure affects the degradation of solid waste in both the lab and the field. Successful implementation of biocells will eliminate the need of land for new landfills and the cost of post-closure monitoring, and will produce energy. They have the potential to revolutionize the management of municipal solid waste.

1.3 Objectives of the Study

The primary objective of this study was to investigate the feasibility of operating landfills as biocells for sustainable waste management. The specific objectives of the study are outlined as follows:

1. To study the effect of inoculum on solid waste decomposition and gas production in biocells by laboratory simulation,
2. To design the biocell for field application,
3. To Install and instrument biocells in the field,
4. To monitor and evaluate biocells in terms of enhanced waste decomposition and gas production.

1.4 Dissertation Outline

The study is divided into seven chapters that are summarized as follows:

Chapter 1 provides an introduction and presents the problem statement and objectives of the study.

Chapter 2 presents the concepts of MSW management, landfills, and biocells, as well as other previously conducted studies related to these topics.

Chapter 3 describes the methodology of the work in the laboratory and field.

Chapter 4 presents the results and analysis of the laboratory study on biocells.

Chapter 5 depicts the construction, instrumentation, and monitoring techniques of two full-scale test cells in field for performance evaluation.

Chapter 6 describes the results and analysis of field test cells and compares them with those of the laboratory experiments to study the feasibility of operating landfills as biocells.

Chapter 7 summarizes the main conclusions from the current research and provides recommendations for future work.

Chapter 2

Literature Review

2.1 Municipal Solid Waste Management

Municipal solid waste (MSW) can be described as leftovers that have no use for the owner, but are a potential source of energy. The US EPA defines MSW as trash or garbage, consisting of everyday items that people use and then throw away, such as product packaging, grass clippings, furniture, clothing, bottles, food scraps, newspapers, appliances, paint, and batteries. Solid waste generation is a by-product of urbanization, rapid industrialization, population growth, and migration (Tahir et al., 2015). According to the Texas Commission on Environmental Quality (TCEQ), MSW can be defined as ”solid waste, resulting from or incidental to municipal, community, commercial, institutional, and recreational activities, including garbage, rubbish, ashes, street cleanings, dead animals, abandoned automobiles, and all other solid waste other than industrial solid waste.” In the USA, about 254 million tons of waste were generated in 2013, of which 87 million tons was recycled and composted (34.3%). Approximately 4.40 pounds of waste was generated per person per day, of which 1.51 pounds per person per day was composted or recycled, as showed in Figure 2-1 (US EPA, 2013).

According to the US EPA (2013), organic materials were the largest component of MSW in the USA, where food waste amounted to 14.6% of the total, paper and paperboard accounted for 27%, and yard trimmings contributed 13.5%. Plastics comprised about 13 %; metals, rubber, leather, and textiles accounted for 9%; wood followed at around 6%; and glass at 5%. Other miscellaneous wastes made up approximately 3% of the MSW generated in 2013 (Figure 2-2). In contrast, the annual waste generation in East Asia and the Pacific region is approximately 270 million tons per year. This quantity is mainly influenced by waste generation in China, which makes up

70% of the regional total. Per capita waste generation ranges from 0.44 to 4.3 kg per person per day for the region, with an average of 0.95 kg/capita/day (Hoornweg et al., 2013).

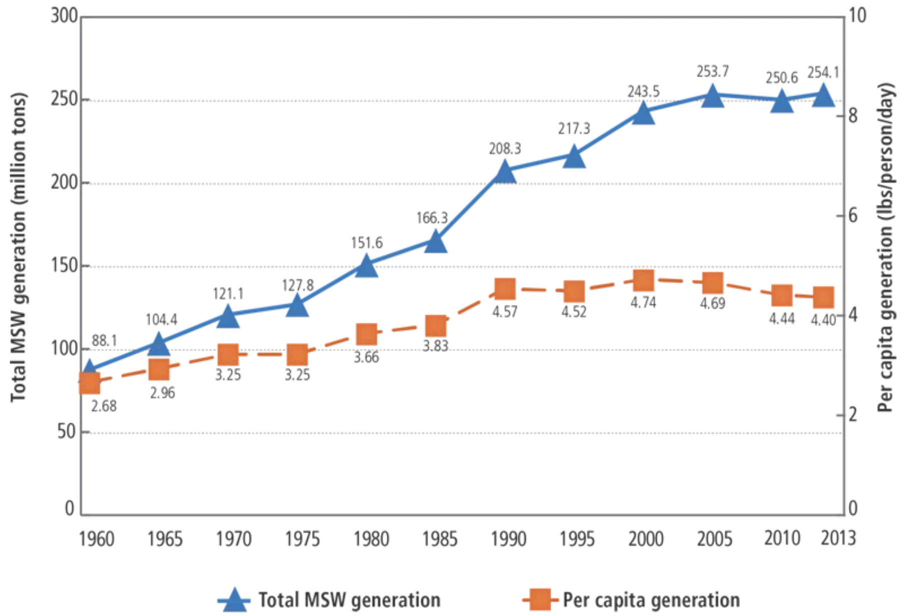


Figure 2-1 MSW generation rates in USA, 1960-2013 (US EPA, 2013)

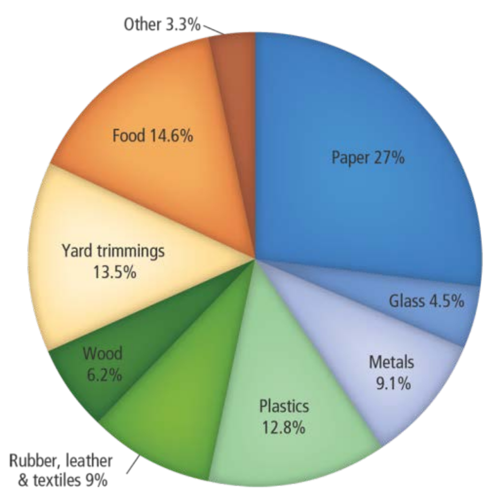


Figure 2-2 MSW composition, 2013 based on 254 million tons (before recycling) (US EPA, 2013)

According to Tahir et al (2015), solid waste generation and management is a burning issue all over the world. It contributes 3% of the total greenhouse gas emissions globally, which impact global warming and climate change. The most common problems associated with improper management of solid waste include disease transmission; odor; nuisance; atmospheric, land, and water pollution; fire hazards; aesthetical nuisance; and economic losses (Yeny and Yulinah, 2012). Both developed and developing countries face the problems associated with solid waste generation and its management. The global population rose to 6.9 billion in 2010, with the majority of people living in developing countries. A major challenge is how to manage the ever-increasing amount of waste generated, especially in developing countries already lacking a sufficient public service infrastructure to manage municipal waste, and where poverty and unplanned settlements lead to unmanaged waste (World Bank, 2012). The increase in solid waste generation rates and its heterogeneous nature create numerous problems related to waste management strategies and their effects on the environment and human health. The waste management hierarchy refers to the five r's: reduction of waste at source, reuse of products, recycling of materials, recovery of energy, and residual management (Vesilind et al., 2002). Waste management practices related to waste generation, recycling, and disposal techniques have evolved substantially over the last few decades. Although the recycling rate increased from 5.6% in the 1960's to 34.7% in 2011 in the USA, landfilling is still the most dominant waste management practice in the USA (USEPA, 2011).

2.2 Landfills

Solid wastes are mainly disposed of in a landfill because it is the simplest, cheapest, and most cost-effective method of disposing of waste (Barrett and Lawler, 1995). In most developing nations with a low-to-medium-income population, almost all of the waste goes to landfills. Even in many developed countries, most solid waste is landfilled. For instance, within the European Union, although policies of reduction, reuse, and diversion from landfills are strongly promoted, more than half of the member states still send more than 75 percent of their waste to landfills (e.g. Ireland sends 92 percent). In1999, landfills were still by far the main waste disposal option for Western Europe (EEA, 2003). When MSW is landfilled without pretreatment, emissions (leachate and biogas) arise during landfill operations and continue after closure (Białowiec, 2011). Leachate is the polluted liquid produced as a result of rain or other water percolating through waste that is landfilled or dumped. Landfill gas (LFG) is a mixture of gases (predominantly methane and carbon dioxide) produced through microbial activity in anaerobic conditions during the degradation of waste that is landfilled or dumped (Johannessen, 1999). Leachate and LFG are greatly influenced by biological processes that take place in the landfill. Based on approximating a highly compacted landfill in Central Europe that has an annual precipitation rate of 600-750 mm, and depending on waste composition, climatic conditions, etc., leachate and LFG will contain approximately 150 (ranging from 70 to 300) m^3 biogas/Mg MSW (based on dry weight) and about 5 $m^3 \cdot ha^{-1} \cdot d^{-1}$ of highly polluted leachate. (Białowiec, 2011). The largest anthropogenic source of atmospheric methane in many developed countries is from landfill emissions. For example, LFG represents 30%, 24%, and 25% of the anthropogenic emissions of methane into the atmosphere in Europe, the United States, and Canada, respectively (Nikiema *et al.,* 2007). One obvious option for managing landfill gas is to capture the produced biogas and use it as an energy source. However, this is viable only when a sufficient quantity of gas is available during the landfill process. Landfills can be categorized as sanitary landfills, conventional landfills, dry tombs, or landfill bioreactors. A biocell is a further improvement of the landfill bioreactor concept (Table 2-1).

Table 2-1 Comparison of major components of landfill (Reconstructed from Hsiao, 2001)

Character	Conventional Landfill/Dry Tomb	Bioreactor Landfill	Biocell
Liner system	Yes	Yes	Yes
Gas collection	Yes	Yes	Yes
Leachate collection system	Yes	Yes	Yes
Monitoring system	Yes	Yes	Yes
Gas production control	No	Yes	Yes
Pre-treatment of MSW	No	Yes	Yes
Leachate recirculation	No	Yes	Yes
Decomposition acceleration by adding substrate	No	No	Yes
Lifetime (Active Period)	50-100 years (Crawford and Smith, 1985)	5-10 years (Pacey et al., 1999)	-

2.2.1 Conventional Landfill

The conventional landfill, also known as the "sanitary landfill," was invented in England in the 1920s. At a landfill, the garbage is compacted and covered at the end of every day with several inches of soil. Landfilling became common in the United States in the 1940s; by the late 1950s, it was the dominant method for disposing of municipal solid waste (Encyclopedia.com). A sanitary landfill is an engineered disposal facility designed, constructed, and operated in a manner that minimizes impacts to public health and the environment. In contrast to open dumpsites and controlled dumps, sanitary landfills undergo thorough planning from the selection of the site up to post-closure management. Thus, although it requires substantial financial resources, it is the most desirable and appropriate method of final waste disposal on land (UNEP, 2005). However, sanitary landfill cannot control the issues associated with landfill gas emissions and land space (Yuen, 2001) due to the slow biodegradation of MSW. Figure 2-3 shows the gas production from conventional and bioreactor landfills.

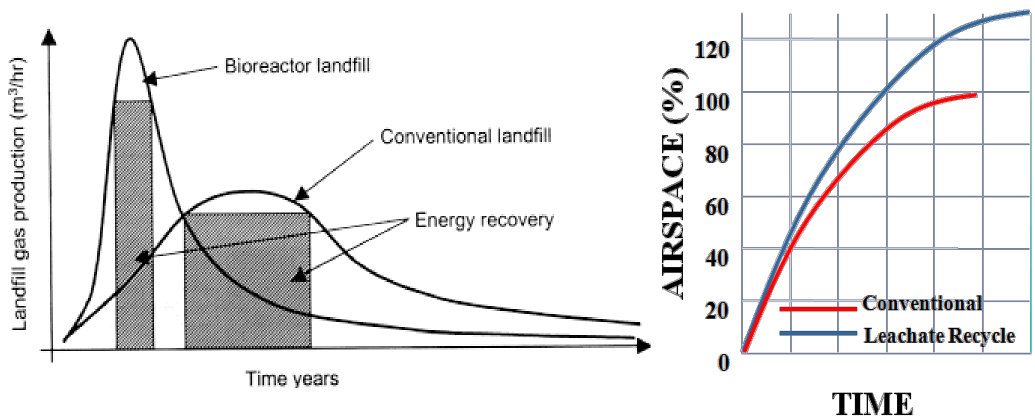


Figure 2-3 Gas production from conventional and bioreactor landfills (Bakas et al., 2011)

2.2.2 Bioreactor Landfill

The concept of a bioreactor landfill was developed in the early 1970's in an effort to overcome the extensive land space requirements of landfills caused by the slow biodegradation of MSW,. According to Reinhart et al. (2002), the benefits of a landfill bioreactor operation were first proven through laboratory studies during the 1970's and pilot and full-scale demonstrations in the 1980's. According to the Solid Waste Association of North America (SWANA), a bioreactor landfill can be classified as "a controlled landfill or landfill cell where liquid and gas conditions are actively managed in order to accelerate or enhance bio-stabilization of the waste." It uses enhanced microbiological processes to transform and stabilize moderately decomposable organic waste within 5 to 10 years, which has enormous environmental, regulatory, monetary, and social benefits (Pacey et al., 1999). The increase in waste degradation and stabilization is achieved through the addition of liquid and air to enhance the microbial processes. Landfill bioreactors are designed and operated to enhance the biodegradation process by increasing waste moisture levels within the landfill (Reinhart and Townsend, 1997). The goal is to achieve the optimum biostabilization of waste. Biostabilized waste does not generate leachate or landfill gas in the quality and quantity that will cause a threat to the environment or human health (Perera, 2005). Some of the key benefits of a bioreactor operation include rapid organic waste degradation and stabilization; maximized landfill gas production; increased landfill space reuse due to rapid settlement during the operational period; improved leachate treatment; reduction in greenhouse gas emissions and other environmental impacts; reduction in post-closure care, maintenance, risks, and overall landfilling cost.

2.2.3 Biocell

Bioreactors have proven successful over the past few decades, but a new landfill facility is needed once the active period of bioreactor landfill is over. For the over-populated areas like Asia, Africa, and Latin America, and even for Europe, landfill space is an emerging issue. In North America, regulations require post-closure activities and financial assurance for 30 years after landfill closure, and a state agency may require additional years of care if needed (US EPA, 2001). A variation of the bioreactor, the “sustainable biocell”, or “biocell,” known as the third generation of landfills, was proposed by Hettiaratchi (2007) to address the problem of space.

According to Hettiaratchi et al (2010), the sustainable landfill biocell can simultaneously solve the problems associated with the slow degradation of MSW and space. The biocell involves sequential operation of a landfill cell to produce methane gas during the first stage of anaerobic degradation, promote in-situ composting within the cell footprint during aerobic degradation during the second stage, and enhance landfill mining for resources and space recovery in the third stage (Meegoda, 2013). Non-recovered waste has a high energy content that can be used as refuse-derived fuel, such as an energy pallet. Some lab based studies have also shown the effects of leachate augmentation by adding nutrients (Jayasinghe et al, 2011; Warith, 2001; Alkaabi et al.,

2009) to achieve enhanced biodegradation of waste. The biocell approach has the potential to eliminate the continuous need to allocate valuable land for new landfills, thereby revolutionizing the management of municipal solid waste.

One advantage of the biocell concept is that it is not operated as a permanent facility (Bartholameuz, 2015). However, it requires enhanced degradation of waste to achieve space recovery, which can be accomplished by the addition of a substrate/supplement with leachate before recirculation into the waste mass, accelerating the rate of decomposition and leading to higher methane generation and faster stabilization of waste.

In a bioreactor landfill, the biological activity in a waste cell is enhanced primarily through leachate augmentation. Recirculation of leachate helps the landfill maintain a wet environment, in addition to supplying nutrients required for the biodegradation and eliminating the leachate treatment (Hettiarachchi, J.P.A., 2013). Many researchers consider leachate recirculation alone as a method for increasing the moisture content of waste. This only accelerates the early hydrolysis and acidogenesis stages, which results in a high acid concentration in the leachate (Yuen, 2001). The modifications of leachate before recirculation can be done through nutrient supplementation such as sludge, enzymes, temperature adjustments, pH control etc., that may aid the biodegradation process. Among these techniques, the addition of sludge is shown to be the most common and oldest practice (Jayasinghe et al., 2010). Jayasinghe et al., (2011) proved that the addition of enzymes can increase the lignin degradation of landfilled waste under anaerobic conditions. Laboratory and field scale investigations have also been on enhancing biodegradation by manipulating the leachate before recirculation. Table 2.2 summarizes the various techniques of enhanced degradation employed by various researchers.

Table 2-2 Enhanced biodegradation of MSW

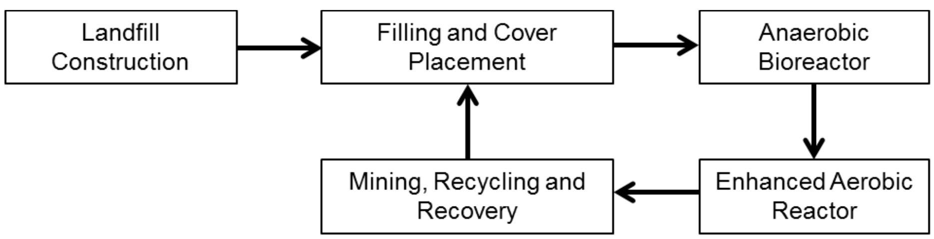
Substrate	Ratio	Feed Stock	Advantage	Reference
Addition of municipal sewage sludge added to the recirculated effluent	5% of the total leachate volume	60% organic waste	25% of landfill air space	Warith (2001)
Addition of sludge, effect of saline water	800 mL of waste- activated sludge + 1400 mL anaerobic digested sludge	60% organic (food), 20% paper, 15% plastics and 5% textile (33.5 to 36.2 kg total)	14% more methane yield	Alkaabi et al. (2009)
Addition of enzyme	0.015% MnP + 0.000069% H2O2	2 g of dried MSW	Degradation of plastics and 36 times higher cumulative methane generation	Hettiaratchi et al. (2011)
Addition of horse manure	6% to total feedstock (118 Mg)	91% Green Waste (1,718 Mg)	20-fold acceleration of methane generation	Yazdani (2010)
Addition of bovine rumen fluid inoculum	15% inoculum	85% MSW	Methane concentration in biogas 42.6%	Lopes et al (2004)
Addition of composted MSW			helps to initiate the methane phase relatively early	Stegmann and Spendlin (1989)

2.3 Biocell Operation

Biocell operation is the advanced form of a bioreactor operation of landfill and involves operating a landfill cell under sequential anaerobic - aerobic conditions with leachate recirculation to take advantage of both forms of biodegradation (Jayasinghe, P., 2013). Leachate augmentation before recirculation by adding nutrients is another technique that can be used to achieve enhanced biodegradation (Jayasinghe et al, 2011; Warith, 2001; Alkaabi et al., 2009). In this section, detail of biocell operation is described.

2.3.1 Sequential Anaerobic-Aerobic Digestion

According to Hettiaratchi (2010), a biocell operates in three stages. During the first phase, it operates as an anaerobic bioreactor, with leachate recirculation and gas extraction for power generation. In the second phase, it operates as an aerobic bioreactor and converts MSW to a compost-like product. The third phase of operation involves mining to recover resources and space, and allows the empty cell to receive waste again so that the cycle can be repeated. This closed loop mode of operation is an attractive alternative to conventional landfilling. Figure 2-4 shows the various phases of operation of a biocell. Details of the stages are described below:



Stages of Landfill	Timeline
Stage I: Anaerobic Bioreactor	5 to 7 years
Stage II: Enhanced Aerobic Reactor	1 to 2 years
Stage III: Mining, Recycling and Recovery	1/2 to 1 year
Total Expected Life Span of the Biocell	6 to 12 years

Figure 2-4 Phases of biocell operation with expected durations (Reconstructed from Hettiarachchi, 2010)

2.3.1.1 Biocell Stage 1: Anaerobic Decomposition

During anaerobic decomposition with gas extraction in a waste cell, the biodegradation rate and landfill gas production depend on temperature, moisture content, and nutrient and organic content in the waste. A higher organic content produces more gas. In a conventional dry-tomb sanitary landfill, it may take as long as 50–100 years to degrade the majority of biodegradable organics (Crawford and Smith, 1985). In the biocell, such degradation is expected to occur within twelve years. Although the concept of bioreactor landfilling is relatively new, there are a number of anaerobic bioreactors being used in Northern America and Europe (Hettiaratchi, 2006). The gas production rate of the Calgary biocell from November 2006 to September 2008 is shown in Figure 2-5. Gas extraction started in October 2006, but was only intermittent, as the facility was being tested and commissioned. The initial gas extraction rate was approximately 150 m^3/h of landfill gas; thereafter, the rate decreased gradually and reached a steady state flow rate of 100 m^3/h throughout the winter.

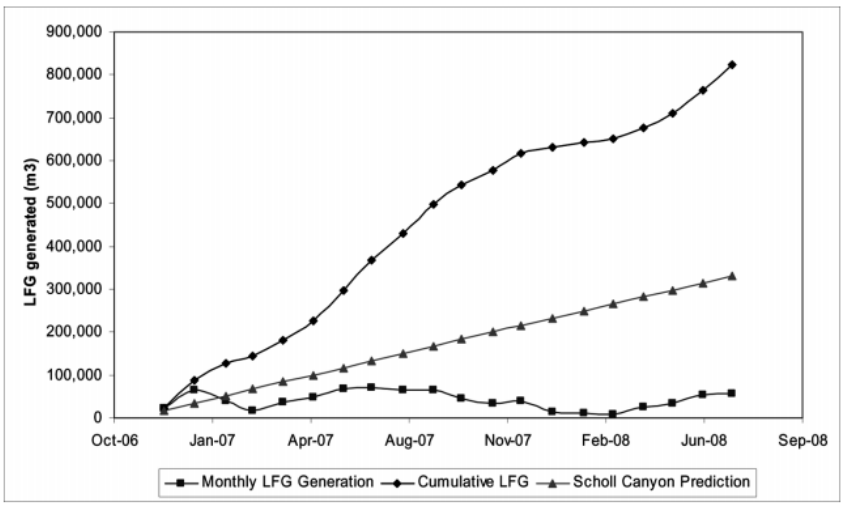


Figure 2-5 Comparison of gas production data at the biocell (without enzyme addition) and a conventional landfill (Hunte, 2010)

For comparison purposes, the Figure 2-5 includes the cumulative landfill gas generation curve for a landfill cell containing 47,000 tons of waste and operated as a dry-tomb sanitary landfill. The Scholl Canyon model (Intergovernmental Panel on Climate Change (IPCC), 1997) was used to generate this curve, using the parameter values; k = 0.016 $year^{-1}$ and Lo = 100 kg of CH_4/ton of waste deposited. The parameter values are those proposed by Environment Canada for a typical sanitary landfill in the province of Alberta

(Thompson et al., 2005). The gas extracted at the biocell was approximately 2.5 times greater than that expected to be produced by a dry tomb type sanitary landfill of similar size and configuration located in Calgary, Alberta. Considering that a gas extraction system is not 100% efficient in collecting the gas produced, in reality, the ratio of gas collected in the biocell to that collected from a dry tomb landfill would be even higher. If 80% collection efficiency is assumed, then the biocell produces three times as much gas as a dry tomb landfill of the same size (Hettiaratchi et al, 2010).

Table 2-3 Case study of anaerobic bioreactors (EPA, 2007)

Location	Footprint	Feedstock	Treatment	Settlement and Gas Production
Crow Wing County (CWC) Landfill, Minnesota	14.1 acres	50,000 tons of MSW annually	Four million gallons of treated and untreated leachate were injected via horizontal, laterals, working face spray, and spray on yard waste composting over intermediate cover	20% settlement in five years
Burlington County bioreactor, New Jersey	10 acre	One million tons of waste		44% increase in effective density of waste
The New River Regional bioreactor, Florida	10-acre	One million tons of waste	24,600 m^3 (6.5 million gallons) leachate was recirculated.	Greatest depth of settlement at injection well and declined with radial distance from the well up to 15 m
Salem County bioreactor, New Jersey	5-acre		0.167 m^3/ton wastes (44 gallons/ton wastes) leachate recirculated.	Rate of settlement was about 1.5 m (5 ft.) per year

2.3.1.2 Biocell Stage 2: Aerobic Decomposition

According to Hettiaratchi et al (2010), once methane production decreases to a critical level, the next stage of the biocell, aerobic treatment, will be initiated. The use of an air injection will rapidly enhance the degradation of the remaining organic waste. Since aerobic degradation occurs at a high rate, this stage may take only a year or two to complete. To convert from anaerobic to aerobic conditions, air has to be introduced to the biocell and maintained to enhance the rate of waste decomposition (Stessel and Murphy, 1992; Hettiaratchi, J.P.A., 2006). The aerobic bioreactor concept was first proposed by Merz and Stone (1962), and there are more than 20 operating aerobic bioreactors in North America. Table 2-4 details various case studies of aerobic bioreactors in the USA.

Table 2-4 Case study of aerobic bioreactors

Location	**Footprint**	**Feedstock**	**Treatment**	**Settlement and Gas Production**	**Reference**
Williamson County bioreactor, Tennessee	7 acres	70,000 tons of waste	3,785 m^3 (one million gallons) leachate, storm water, and air were injected into vertical risers with force	5.1–10.7% decrease in waste height over a 59-month period of operation	EPA (2007)
Columbia County landfill, Georgia			The air injection rates were 56 m^3/min	The greatest settlement was 9%. The methane generation was reduced to 50% after aeration started.	Hudgings and Harper (1999)
Privately operated bioreactor, Atlanta, Georgia			The air injection rates were 100 m^3/min	The greatest settlement was 10%. The methane generation was reduced to 50 - 90% after aeration started.	

The gas extraction system used during the anaerobic stage is used to pump air into the landfill to create aerobic conditions. The recirculation of appropriately adjusted leachate is also required. Towards the end of aerobic decomposition, periodic testing of waste from boreholes and the analysis of the leachate ensures complete biodegradation. Partially degraded waste can cause problems in post mining; therefore, prior to moving to the mining step, it is essential to ensure that most of the organic waste is decomposed. To accomplish this, the aerobic conditions may have to be maintained for a longer time period than desired. The aerobic stage enhancement of a waste cell can be achieved by controlling the biocell temperature, augmenting the leachate, and bioventing (Ishigaki et al., 2003). Aerobic composting enhancement techniques, such as inoculating microbes and seeds, and adding mature compost can also be adopted if experimentally proven (Cayuela et al, 2010; Shin et al., 1999; Yen et al., 2006).

2.3.1.3 Biocell Stage 3: Mining for recovery of useful/recyclable products

In the third stage of a biocell, even with enhanced biodegradation, there will be some non-recoverable residual waste which has high calorific value. These wastes may include textiles, wood, and fractions of other organic waste types. These can be used to produce refuse-derived fuel (RDF) and used for co-incineration in cement kilns (Hettiaratchi et al, 2010). Once these materials are mined out, there will be sufficient space in the biocell begin the process anew.

2.3.2 Inoculum and Nutrient Addition: Effect on Waste Degradation

2.3.2.1 Addition of Enzymes

The manipulation of leachate with enzymes before recirculation was first studied by Lagerkvist and Chen (1993). The effect of enzymes on waste degradation was studied separately during acidogenic and methanogenic degradation stages. It was observed that adding enzymes enhanced the degradation of waste to a greater degree than the control cell. Success of this manipulation was measured by cellulose content and conversion of volatile solids (VS). The observed conversion of cellulose was 42-70% in cells with added enzymes and 29% in cells without added enzymes. The conversion of VS was approximately 40% to 50% in enzyme-added cells. Cirne et al. (2008) suggested that the addition of cellulolytic enzymes under anaerobic conditions resulted in better performance in terms of degree of solubilization, with an approximate 34% increase in the degree of solubilization. Pre-digested organic waste (800 g, 22.8% TS and 22.1%

VS) was used, and the enzymes used were commercial preparations of cellulase (Celluclast 1.5L™, β- glucosidase, and Novozyme 188™). According to Bisaria (1991), both bacteria and fungi can produce cellulases for the hydrolysis of lignocellulosic materials. These microorganisms can be aerobic or anaerobic, mesophilic or thermophilic. Bacteria belonging to Clostridium, Cellulomonas, Bacillus, Thermomonospora, Ruminococcus, Bacteriodes, Erwinia, Acetovibrio, Microbispora, and Streptomyces can produce cellulases.

Many microorganisms are capable of degrading and utilizing cellulose and hemi-cellulose as carbon and energy sources; lignin is highly resistant to degradation (Higuchi, 2006). The most recent study of adding enzymes in a biocell was conducted using partly degraded MSW samples collected from the 30-year old City of Calgary landfill to determine the feasibility of augmenting leachate with different peroxidase enzymes to increase the rate of waste degradation during later stages of anaerobic landfill bioreactor operation. For this study, three types of peroxidase enzymes, lignin peroxidase (LiP), manganese peroxidase (MnP), and soybean peroxidase (SbP) were selected to evaluate their ability to further degrade partially degraded MSW (Jayasinghe et al., 2011). The lab experiment results showed that the enzyme MnP performed best in terms of yielding methane for lignin-rich degraded MSW (Figure 2-6 and Figure 2-7).

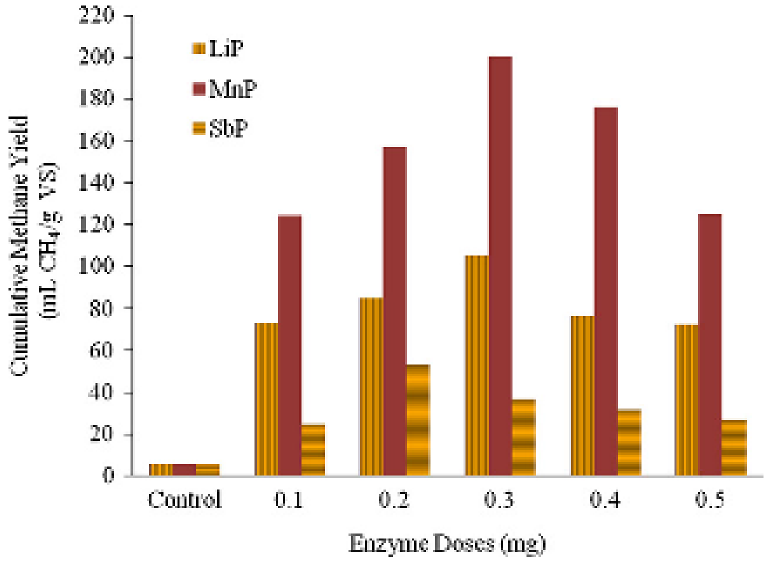


Figure 2-6 Effect of enzyme type on methane yield at different levels of enzyme dose at 0.0046 enzyme: H_2O_2 ratio (Jayasinghe et al., 2011)

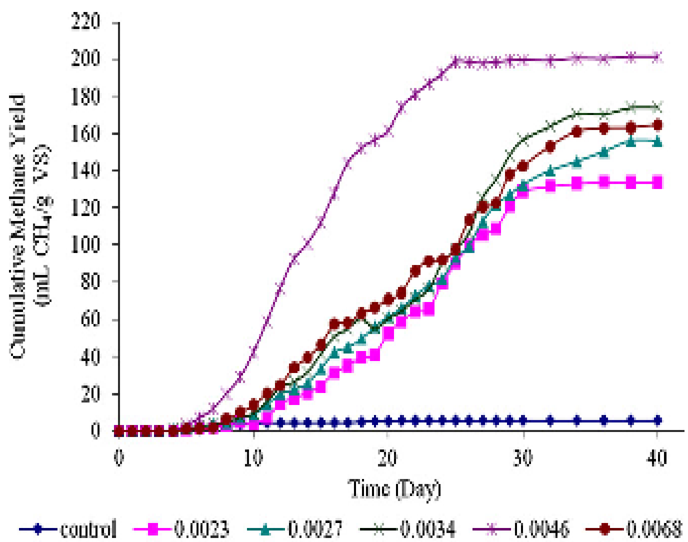


Figure 2-7 Cumulative methane yield for MnP at different enzyme: H_2O_2 ratios at 0.3 mg enzyme dose (Jayasinghe et al., 2011)

2.3.2.2 Addition of Manure

In California's Yolo County, a full-scale study was done to demonstrate that the anaerobic-controlled bioreactor landfill represents a cost-effective route for the recovery of methane from solid waste. A landfilled-based two-stage (anaerobic/aerobic) batch digester cell was constructed, operated, and monitored for treatment of source-separated green waste while recovering energy and compost. The performance of this unique digester was evaluated in terms of cell operating temperature, leachate quality, methane generation rate, air emissions, waste decomposition indicators, energy production, and compost quality. The decay rate observed in the landfill digester (k = 0.82/yr) represents about a 20-fold acceleration of methane generation compared to the U.S. EPA default for solid waste (Yazdani, 2010). In a study in New Mexico, different combinations of feedstock were tested in the lab to see the effects of co-digestion with cow manure, as shown in Table 2-5.

Table 2-5 Experimental results for different types of feedstock (Macias-Corral et al., 2008)

Parameter	**MSW**	**CM**		**OFMSW + CM**
		Run 1	Run 2	
Duration of experiment (days)	113	73	45	141
Total biogas produced (m^3)	4.0	64.8	17.1	96.6
Average methane content (%)	73.1	72.3	72	73.0
Methane produced (m^3 standard temperature and pressure)	2.9	41.8	12.3	54.0
Methane yield (m^3 CH_4/ton dry waste)	37	62	66	87
Methane yield (m^3 CH_4/kg VS)	0.03	0.08	0.07	0.10

The organic fraction of municipal solid waste represented about 61% of the total MSW and was composed of approximately 62% paper, 23% food waste, and 15% yard clippings. Four hundred and fifty (450) liters of tap water were added to the solid phase, and the resulting composition was 63.7% paper, 18.2% food waste, 9.1% grass clippings, and 9% cow manure. The experimental results showed that co-digestion had higher methane gas yields than single waste digestions. In addition, co-digestion of organic fraction of MSW (OFMSW) and cow manure (CM) promotes synergistic effects resulting in higher mass conversion and lower weight and volume of digested residual (Macias-Corral et al., 2008).

2.3.2.3 Addition of Nitrogen and Phosphorus

Warith et al. (1999) conducted a pilot scale experimental study with simulated landfill cells filled with MSW over a period of 65 weeks to study the effects of adding supplemental materials to leachate during recirculation. The recirculated leachate was supplemented in two ways: by adding nitrogen and phosphorus with a buffer as supplemental nutrients to balance the nutrient deficiency within the solid waste matrix, and by adding primary sludge to increase the microbial population within the waste. The effectiveness of adding these materials was determined by analyzing the characteristics of effluents, such as BOD, COD, TOC, and heavy metal concentrations. Experimental results indicated that the addition of supplemental materials to leachate during recirculation significantly enhances the rate of biodegradation of solid waste.

2.3.2.4 Addition of Sludge

Anaerobically digested sewage sludge is a great source of microorganisms, nitrogen, phosphorous, and other nutrients (Warith 2005). A laboratory study carried out by Alkaabi et al. (2009) showed that the addition of sludge under saline conditions enhances the biodegradation of MSW. Two groups of laboratory scale bioreactor cells were used. One group was used to study the effect of the salinity of water on waste degradation under different operating conditions, and other group was used to study the impact of adding sludge under saline conditions. The methane yield was about 14% more in the bioreactors that added sludge at different salt concentrations than in the bioreactors that were operated without the addition of sludge. Barlaz et al (1990) experienced carboxylic acid accumulations and decreases in pH when sludge was added to fresh MSW. The results of their study confirmed that adding sludge without buffer addition does not stimulate methane production. Gulec et al., 2000 showed that in 10 L laboratory-scale batch digesters filled with two-year old MSW at ratios of 1:9, 1:6 and 1:4 (anaerobically digested sludge to waste on wet basis), the pH of leachate ranged from 7.0 to 8.5. The acidic range in the control reactors (no addition of sludge) experienced a sharp drop in pH levels. This study suggested the buffer capacity of sludge. In addition, Christensen and Kjeldsen (1992) reported that if the anaerobic conditions are already established, the addition of sewage sludge to MSW might have a limiting effect on waste biodegradation.

2.3.3 Advantage of Biocell

2.3.3.1 Enhanced Methane Production

A lab scale study of the Calgary biocell, conducted by Jayasinghe (2013), showed degradation of plastics and 36 times higher generation of cumulative methane due to the addition of enzymes to leachate. Another study conducted by Yazdani (2010) showed a 20-fold acceleration of methane generation due to the addition of manure to green waste in a biocell in Yolo County, California.

2.3.3.2 Accelerated Stabilization of Waste

Due to the aeration in the later stage, waste stabilization occurs faster in biocells because of the degradation of lignin in the waste mass (Hettiaratchi, 2014). According to Bartholameuz (2015), an air injection, using vertical piping systems, showed the best degradation rates and had the highest degree of aerobic activity. In the field scale operation of the Calgary biocell, the initial volume of the cell was 74,008 m^3, with a total of approximately 47,900 tons of waste. As shown in Figure 2-8, after 1000 days of post-closure operation, the total strain at the peak of the biocell was approximately 16%, and the volumetric strain was approximately 10% (Hunte, 2010).

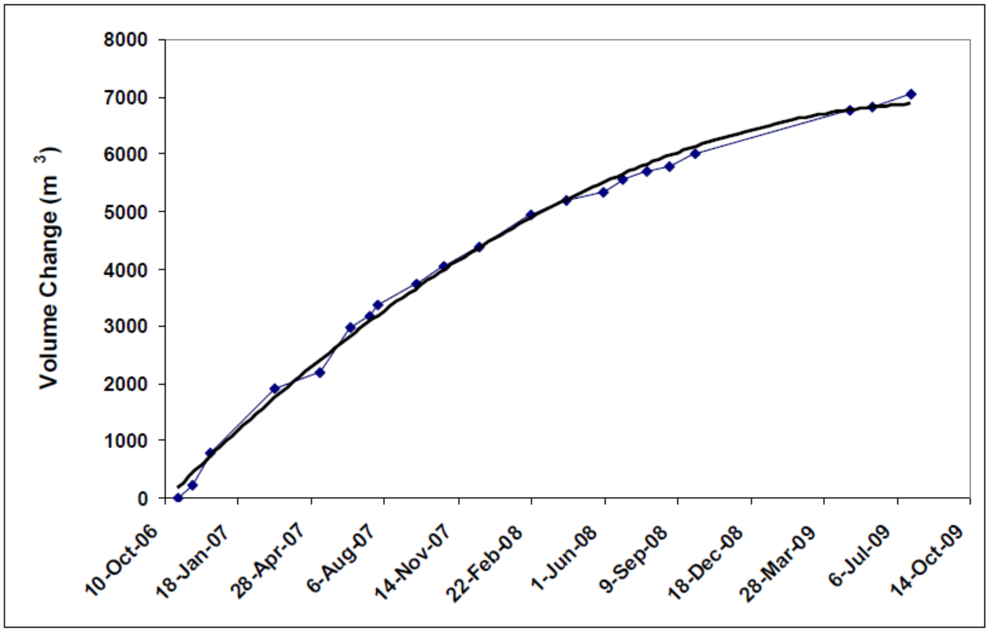


Figure 2-8 Change in Calgary biocell volume over monitoring period (Hunte, 2010)

2.3.3.3 Recovery and Reuse of Space

Once waste stabilization has been attained, a rapid aeration process can help prepare the waste cell for mining. In the lab, increasing the aeration rate ten times resulted in increasing the rate of evaporation of waste by three times. Dry waste is considerably less harmful, has reduced odors, and less mobile toxic compounds (Bartholameuz, 2015). The ability to reuse space in a biofill is a great benefit for areas in or near urban centers where it is difficult to locate space for new landfills (Hunte, C., 2010).

2.3.3.4 Improvement of Leachate Quality

Blakey et al. (1997) and Viste (1997) found in their field studies that adding biosolids to waste resulted in a modest increase of biogas production and improvement in the quality of leachate. This offered a considerable cost advantage since secondary treatment was not required (Read et al., 2001).

2.4 Waste Decomposition in Landfills

Landfills are considered as heterogeneous systems due to their variable refuse characteristics. Furthermore, placement methodology, hydrological conditions, compaction, and seasonal variations make the system more complex and difficult to predict. Stratification of refuse can occur in lifts and localized volumes. Key parameters controlling degradation are refuse composition, moisture content, temperature, redox conditions viz. Eh and pH, hydraulic gradients, xentiobiotics, metals, and oxic-anoxic interfaces. Christensen and Kjeldsen (1989) observed rapid depletion of oxygen in landfills and reported complete depletion in a timeframe of a week, after which nitrate was consumed rapidly. With depletion of oxygen, the anaerobic environment enables dominance of facultative and then obligates anaerobes. Numerous interacting microbial species use a variety of substrates and intermediates such as nitrate, sulfate, and carbon dioxide, theoretically in sequence of available energy from species' selective electron donors. Hence, bacterial populations can be a good indicator of the degree of degradation in a particular lift. Redox conditions dictate availability of electron donors, and the species deriving maximum energy often gains a kinetic advantage over the others. Mixed cultures coexist due to five complex transport phenomena in a landfill matrix. Commonly reported species are Clostridium butyricum, C. pectinovorum and C. fulsincum for pectin dissimilation, C. thermocellum and C. cellobiopavum, and C. cellulosae dissolvens for cellulose degradation (CRC Press Inc., 1990). A study of anaerobic bacterial counts indicated that total anaerobes ranged from 103 cells per dry gram in cover soil to 109 in grass, food waste, and fresh refuse. Hemicellulolytics ranged from 160 cells per dry gram in cover soil to 109 in grass. The highest cellulolytic population was measured on branches (316 cells per dry gram), while the maximum acetogenic population was 104, measured on leaves. The highest methanogen populations were measured on leaves (103) and one of two fresh refuse samples (105) (Qian and Barlaz, 1996).

2.4.1 Aerobic Decomposition

During aerobic decomposition, organic components presented in the MSW mass are oxidized in the presence of aerobic bacteria to produce carbon dioxide and water vapor (Themelis and Ulloa, 2007). Aerobic decomposition of waste takes place soon after waste disposal when the organic portions undergo biochemical reactions for a short period of time due to the presence of oxygen in the voids of waste. The reaction duration depends on the amount of available oxygen in the waste, which primarily depends on the compaction effort in the MSW disposal. Loose compaction results in more porous waste, which stores a greater amount of oxygen than high compaction. Once the aerobic decomposition is completed, the biochemical reaction of the waste shifts to the anaerobic stage because of the breakdown of oxygen in the waste.

2.4.2 Anaerobic Decomposition

In anaerobic decomposition, MSW is converted to methane and carbon dioxide in the presence of anaerobic microorganisms by a series of chemical conversion processes. In landfills, anaerobic digestion is the main bio reaction of waste, and it takes place in three stages. In the first stage, the complex organic matter or polymer is broken down to soluble monomer molecules by fermentative bacteria. In the second phase, these monomers are converted by acid-forming bacteria to simple organic acids, such as acetic acid, propionic acid, butyric acid, ethanol, carbon dioxide, and hydrogen. In the final stage, methane is formed either by breaking down the acids to methane and carbon dioxide, or by reducing carbon dioxide with hydrogen in the presence of methanogenic bacteria (Barlaz et al., 1990; Christnensen and Kjeldesn 1989; Themelis and Ulloa 2007).

2.4.3 Phases of MSW Decomposition

A number of studies have been carried out on the biodegradation of waste in the landfills. Various researchers (Christensen and Kjeldsen, 1989; Barlaz, et al., 1989) have characterized the stabilization of waste in terms of an idealized sequence of phases between the disposed of fresh MSW and well-decomposed waste. The phases of the MSW biodegradation process have been reviewed by many researchers, such as Warith (2003), Warith et al. (2005), White et al. (2004), Zacharof & Butler (2004), Barlaz et al. (1989), Al-Kaabi (2007), Kjeldsen et al. (2002), and Christensen et al. (1989). Biodegradation of waste can be divided into five distinct phases (Warith et *al.,* 2005), as shown in Figure 2-9.

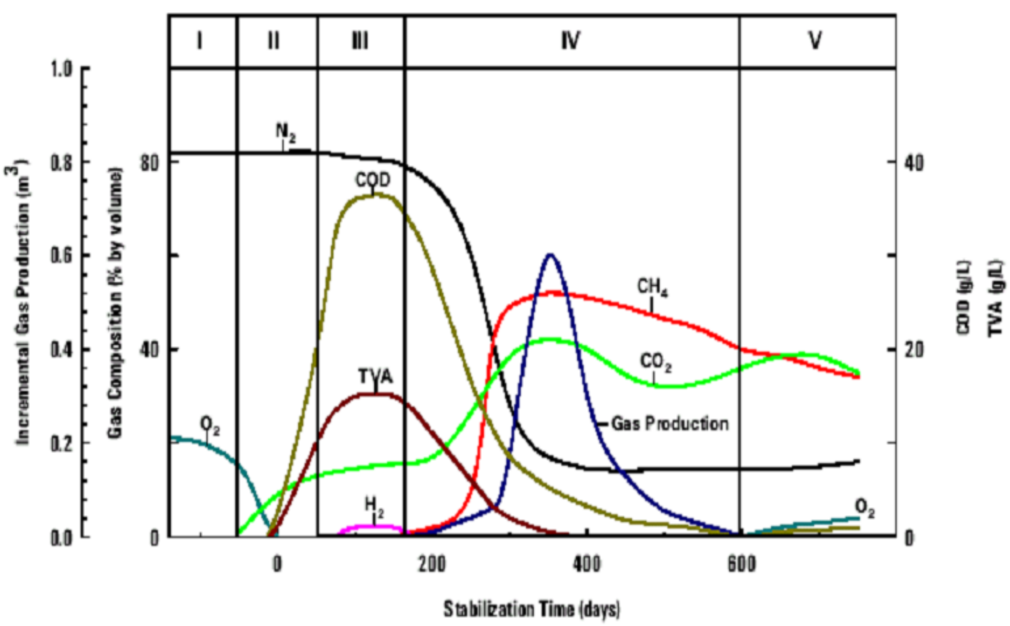


Figure 2-9 Phases of degradation in a typical landfill (WMI, 2000)

Rate and characteristics of produced leachate and biogas vary from one phase to another, reflecting the microbially-mediated processes taking place inside the landfill waste (Reinhart and Al-Yousfi, 1996). The phases of decomposition of waste experienced by Warith et al. *(*2005) are described below.

Phase I: Aerobic phase

After waste is deposited in the landfill, the level of carbon dioxide and heat-produced temperature rises to approximately 30 degrees Fahrenheit. Both oxygen and nitrate are consumed, with the soluble sugars serving as the carbon source for microbial activity. The quantity of oxygen in the waste usually depends on the compaction level. This phase is also associated with accumulation of moisture within landfills. An acclimation period (or initial lag time) may be observed until sufficient moisture develops and supports the growth of the microbial community (Reinhart and Al-Yousfi, 1996). The timeframe of this phase may vary from months up to one year.

Phase II: Transition phase

In this phase, a transformation occurs with the breakdown of oxygen that is trapped within the pores of waste, and the anaerobic microorganisms become active. After the onset of anaerobic conditions, the carbon dioxide dissolves and numerous organic acids are produced, resulting in the production of acidic leachate. The primary components of waste organic matter are carbohydrates, lipids, and proteins. In this stage, these components are broken down sequentially by cellulolytic, lipolytic, and proteolytic bacteria, into soluble monomers such as soluble sugars, amino acids, long-chain carboxylic acids, and glycerol (Barlaz et al. 1990) via hydrolysis (NAS, 1977). In hydrolysis, covalent bonds are split in a chemical reaction with water. By the end of this phase, measurable concentrations of COD and volatile organic acids can be detected in the leachate (Reinhart and Townsend 1998). The timeframe of this phase may vary from one to two years. Bacteria of genre Bacteroides, Lactobacillus, Propioni-bacterium, Sphingomonas, Sporobacterium, Megasphaera, and Bifidobacterium are most common in this phase and include both facultative and obligatory anaerobes (Deublein and Steinhauser, 2008).

Phase III: Acid formation phase

In the acid formation phase, acid-forming bacteria (acidogens) become active. The acidogens include both facultative and obligate anaerobic fermentative bacteria, including Clostridium spp., Peptococcus Anaerobus, Bifidobacterium spp., Desulphovibrio spp., Corynebacterium spp., Lactobacillus, Actinomyces, Staphylococcus, and Esherichia coli (Metcalf & Eddy, 2004). In the first stage of this phase, acidogens convert the soluble monomers into short-chain organic acids (volatile fatty acids with C>2, such as lactic, propionic, and butyric acids) (Khanal, 2008). Alcohol, carbon dioxide (CO_2), and hydrogen (H_2) are also produced in this stage. In the second stage of this phase, the obligate proton-reducing acetogens become active. Acetogenic microbes convert the volatile fatty acids and alcohol formed in the first stage into acetic acid (CH_3COOH) or acetate (CH_3COO-), H_2, and CO_2. The conversion of short-chain carboxylic acids to acetate is only thermodynamically favorable at very low hydrogen concentration. In an active anaerobic ecosystem, however, there is a hydrogen-scavenging population, i.e. methanogens. If fermentative and methanogenic activities are not balanced, intermediates will accumulate and may percolate from the landfill as leachate (Barlaz et al. 1990). The overall process of this phase is as follows:

$$C_6H_{12}O_6 \rightarrow 2C_2H_5OH + 3CO_2$$

$$C_2H_5OH + H_2O \rightarrow CH_3COOH + 2H_2$$

Phase IV: Methane fermentation phase

During this phase, both methanogens and sulfate-reducing bacteria become active in the degradation process. The hydrophilic methanogenic bacteria, which is strictly anaerobic, uses the acetic acid/acetate from Phase III and forms methane and carbon dioxide. Acetotrophic (also called acetogenic or aceticlastic) methanogens, including bacteria from the genres Methanosarcina, Methanosaeta, Methanobacterium, Methanobacillus, Methanococcus, etc., perform this conversion (Khanal, 2008). Methane constitutes approximately 50-60% (by volume) of landfill gas composition (Barlaz, et al., 1990; Warith and Sharma, 1998). The conversion of the acetic acid to the gaseous products CH_4 and CO_2, reduces the oxygen demand (BOD, COD) and increases the pH of the remaining waste, thereby removing heavy metals by precipitation. This phase can be as long as 30 years in a conventional landfill; however, in a bioreactor landfill and biocell, the process can be reduced to 3-5 years with leachate recirculation. The overall process of this phase is as follows:

$$CH_3COOH \rightarrow CH_4 + CO$$

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

Phase V: Maturation phase

In this phase, gas production drops significantly, the leachate pH stays steady in a slightly alkali phase, biodegradable organic matter is stabilized, and volatile organic content and nutrients decrease. Concurrently, there is an increase in the rate of cellulose and hemicellulose hydrolysis. MSW degradation can take from 30 to 100 years in a traditional landfill. Leachate recirculation in a bioreactor landfill, however, accelerates the whole process, resulting in the potential for higher gas production/recovery and more stable leachate during the subsequent methane fermentation phase.

2.5 Components of Waste Degradation

2.5.1 Landfill Leachate

Landfill leachate is the liquid generated from the waste by excess water percolating through the waste layers in a landfill. Various pollutants in the waste can be transferred by the physical, chemical, and microbial processes from the waste material to the percolating water (Christensen and Kjeldsen, 1989; Kjeldsen et al., 2002). Landfill leachate has harmful environmental impacts, as it pollutes groundwater and surface water upon contact. The most severe environmental impact from landfills is probably the risk of the groundwater being polluted by leachate. Leachate is the byproduct of the waste decomposition process and is the result of precipitation, evaporation, surface runoff, infiltration, and storage capacity in a landfill. The moisture content of solid waste is governed by the occurrence of percolation by precipitation and is the primary cause of leachate production, which transpires as soon as the moisture content of waste goes beyond its field capacity. Other products, such as methane, carbon dioxide, and organic acids, are also produced during the biodegradation process of solid waste. The amount of leachate generated also depends on weather conditions, cover soil characteristics, and vegetation. Many studies have been conducted on landfill leachate (Reinhart, 1996; Rees, 1980; Kjeldsen et al., 2002; and El-Fadel et al., 1997). Several factors affecting leachate generation, such as initial moisture content of waste, amount of water recirculating the waste, climatic conditions, composition and type of waste, and density of waste have been studied by El-Fadel el al., 1997a; and Rees, 1980.

Landfill leachate consists of various organic and inorganic compounds of extracted, dissolved, and suspended materials that may contain pollutants. Leachate usually contains four groups of pollutants: dissolved organic matter, inorganic macro-components, heavy metals, and xenobiotic organic compounds (Kjeldsen et al. 2002).

Various factors such as waste composition, age of the waste, phase of decomposition, temperature, and land filling technology affect the quality and composition of leachate. According to Kjeldsen et al. (2002), the major components of leachate are dissolved organic matter; inorganic macro nutrients such as calcium (Ca2+), magnesium (Mg2+), sodium (Na+), potassium (K+), ammonium (NH4+), iron (Fe2+), manganese (Mn2+), chloride (Cl), sulfate (SO42-), and hydrogen carbonate (HCO 3-); and heavy metals such as cadmium (Cd2+), chromium (Cr3+), copper (Cu2+), lead (Pb2+), nickel (Ni2+), and zinc (Zn2+). The leachate composition of the different phases of biodegradation is shown in Table 2-6.

Table 2-6 Leachate composition for different phases (Kjeldsen et al., 2002)

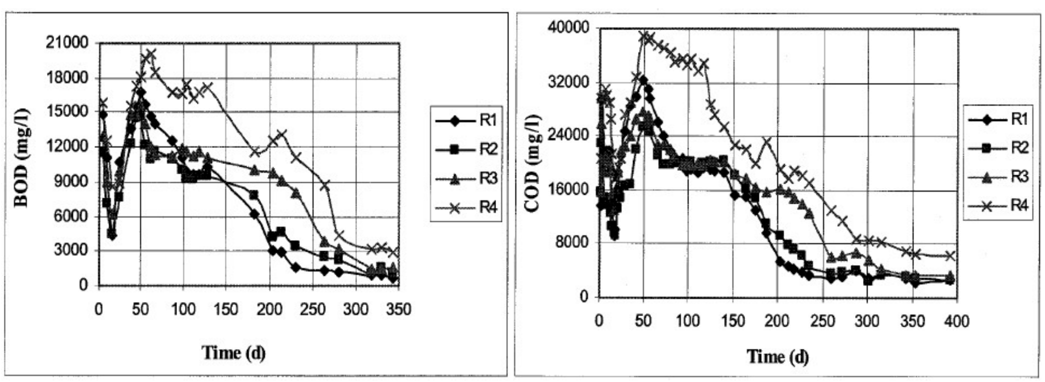
Parameter	**Acidogenic phase**		**Methanogenic phase**		**Average**
	Average	**Range**	**Average**	**Range**	
pH	6.1	4.5-7.5	8	7.5-9	
Biological oxygen demand	13000	4000-40000	180	20-550	
Biological oxygen demand	22000	6000-60000	3000	500-4500	
BOD_5/COD	0.58		0.06		
Sulfate	500	70-1750	80	10-420	
Calcium	1200	10-2500	60	20-600	
Magnesium	470	50-1150	180	40-350	
Iron	780	20-2100	15	3-280	
Manganese	25	0.3-65	0.7	0.03-45	
Ammonia-N					740
Chloride					2120
Potassium					1085
Sodium					1340
Total phosphorus					6
Cadmium					0.005
Chromium					0.28
Cobalt					0.05
Copper					0.065
Lead					0.09
Nickel					0.17
Zinc	5	0.1-120	0.6	0.03-4	

In the acidogenic phase, the pH of leachate is usually at a wider range of 4.5-7.5 than the methanogenic phase. The methanogenic bacteria in MSW produces methane and carbon dioxide gas from hydrogen gas and acid when the pH ranges between 6.8 and 8.0 (Warith, 2003). If the pH value falls outside of the range of 6.7 to 8.0, the biodegradation process and methane gas production slows down. According to Christensen et al. (1996), if the methanogenic activity is limited, the conversion of acetic acid to methane and carbon dioxide decreases, and acids accumulate. This leads the pH value to decrease and may stop or retard the methane production. According to Reinhart and Al-Yousfi (1996), the pH is 5.4 to 8.1 in the transition phase of waste decomposition in a bioreactor landfill, and methane fermentation takes place when the pH is 5.9 - 8.6 (Table 2-7).

Table 2-7 Characteristics of leachate of bioreactor landfill (Reinhart and Al -Yousfi, 1996)

Parameter	**Phase II**	**Phase III**	**Phase IV**	**Phase V**
	Transition	**Acid formation**	**Methane fermentation**	**Final maturation**
BOD (mg l^{-1})	0 – 6,893	0 - 28,000	100 - 10,000	100
COD (mg l^{-1})	20 - 20,000	11,600 - 34,550	1,800 - 17,000	770 – 1,000
TVA (mg l^{-1} as Acetic Acid)	200 - 2,700	0 - 30,730	0 - 3,900	-
BOD/COD	0.1 - 0.98	0.45 - 0.95	0.05 - 0.8	0.05 - 0.08
Ammonia (mg l^{-1} N)	76 - 125	0 - 1,800	32 - 1,850	420 - 580
pH	5.4 - 8.1	5.7 - 7.4	5.9 - 8.6	7.4 - 8.3
Conductivity (µmhos cm^{-1})	2,200 - 8,000	10,000 - 18,000	4,200 - 16,000	-

The BOD to COD ratio is an indicator of the proportion of biologically degradable organic matter to total organic matter. This ratio decreases with the age of the landfill and more degradation products are leached from deposited residues (Reinhart et al., 1998). The acidogenic phase is the early stage of waste degradation and is generally characterized by a ratio of BOD concentration to COD concentration greater than or equal to 0.1 and sulfate levels between 70 and 1,750 mg/L (Reinhart et al., 1998). Typical characteristics of the later phase of waste decomposition, the methanogenic phase, are BOD/COD less than 0.1 and sulfate values between 10 and 420 mg/L (Reinhart et al., 1998). Al-kaabi (2007) measured the BOD and COD variations of leachate on the laboratory scale reactors. In the anaerobic stage, COD concentrations in all bioreactors increased dramatically in the beginning, due to the lack of oxygen and transition to the anaerobic phase, allowing the COD concentration to increase as the hydrolysis continued. This increase was followed by a decrease in COD concentrations in all bioreactors, as a result of an increase in the methanogenic activity, and a subsequent rise in the daily methane production. Al-kaabi (2007) also observed that the BOD concentration increased in all of the bioreactors at the beginning of the anaerobic stage, as a result of low methanogenic activity which facilitated the accumulation of organic acids from hydrolysis and acidogenesis. The author reported that the BOD peak reduction showed that the addition of sludge enhanced the biodegradation of MSW. Figure 2-10 shows the BOD and COD variations of leachate on the laboratory scale reactors, as reported by Al-kaabi, 2007.



(a) (b)

Figure 2-10 (a) BOD and (b) COD variations of leachate on the laboratory scale reactors (Al-kaabi, 2007)

2.5.2 Landfill Gas

Landfill gas or biogas is the main product of waste decomposition in a landfill and is primarily composed of approximately 40 – 60% methane (CH_4), and carbon dioxide (CO_2). Typical composition of landfill gas is shown in Table 2-8. The high methane content is a potential energy source, as well as a greenhouse gas, if it is emitted into the environment. According to the Intergovernmental Panel on Climate Change (2004), methane has 22 times more global warming potential than carbon dioxide over a period of one hundred years. Landfill gas also contains varying amounts of nitrogen, oxygen, water vapor, hydrogen sulfide, and other contaminants. Most of these other contaminants are known as "non-methane organic compounds" or NMOCs and amount to less than 1% of total landfill gas. According to a study on 6,000 MSW landfills in the United States (USA) conducted by the Environmental Protection Agency (EPA), landfills are the largest source of anthropogenic methane emissions in the USA. The USEPA (2008) identified more than 100 trace constituents, including non-methane organic compounds (NMOCs) and volatile organic compounds (VOCs) that were emitted by landfills. The model of the USEPA (2005) User's Guide for Landfill Gas Emissions incorporates default emission factors for 46 trace components.

Table 2-8 Typical composition of landfill gas (Tchobanoglous et al. 1993)

Components	Percentage (%)
Methane, CH_4	45-60
Carbon dioxide, CO_2	40-60
Nitrogen, N_2	2-5
Oxygen, O_2	0.1-1
Sulfides, disulfides, mercaptans, etc.	0-1.0
Ammonia, NH_3	0.1-1.0
Hydrogen, H_2	0-0.2
Carbon monoxide, CO	0-0.2
Trace constituents	0.01-0.6

MSW disposed of in a landfill consists of food, paper, wood, yard trimmings, plastic, textiles and leather, Styrofoam, construction and demolition (C&D) waste, glass, metal waste, and fines. Among these, the biodegradable components such as food, paper, wood, yard trimmings, textiles, and leather are the main contributors of landfill gas, as only the organic fraction of MSW in the landfill decomposes through a series of interacting microbial processes and produces methane (CH_4), carbon dioxide (CO_2), water (H_2O), and several other traces materials. However, the heterogeneity of the MSW, together with the frequently unclear nature of its contents, makes landfill gas production difficult to predict. The rate of methane production depends on several factors, such as the waste composition, landfill geometry, organic contents of the waste, compaction, density, age of waste, pH, particle size, and initial and recirculated water/leachate content, as well as climatic factors such as the annual rainfall and temperature.

Landfill gas production can be estimated by several methods. The most common landfill gas model in the United States is the EPA's landfill gas emissions model (LandGEM), shown in Eq. 1 (USEPA, 1997):

$$\mathrm{Q_M} = \sum \mathrm{k.L_o.M_i}\left(\mathrm{e}^{-\mathrm{k.t_i}}\right) \quad \ldots\ldots\ldots\ldots\ldots\ldots \text{(2.1)}$$

Where,

Q_M = Methane generation rate in the t^{th} year, m^3/year,

Lo = Methane generation potential, m^3/Mg,

Mi = Mass of solid waste place in biocell in the i^{th} year, ton,

k = Decay rate constant, $year^{-1}$

ti = Age of the waste mass in the t^{th} year, year.

Similarly, the total volume of methane that can be produced from a landfill depends on the waste's potential ultimate generation of methane, as represented below:

$$V = L_o.M_i(1 - e^{-k.t_i}) \quad \ldots\ldots\ldots\ldots \text{(2.2)}$$

Where, V= cumulative methane generated until time t (m^3)

The major drawback of this model is that it considers the waste as a single component and doesn’t incorporate the effect of increasing waste mass. To improve the model, USEPA (2005) modified the equation to reflect that that the mass of waste added to a landfill be included for 1/10th of a year:

$$Q_{CH_4} = \sum_{i=1}^{n} \sum_{j=0.1}^{1} k.L_o.\frac{M_i}{10} e^{(-k.t_{ij})} \ldots\ldots\ldots\ldots (2.3)$$

The quantity of methane production and its rate are governed by two factors: the ultimate methane yield (Lo) and the decay rate (k). In enhanced leachate recirculation (ELR) or a bioreactor landfill, the typical value of Lo is 170 m^3/Mg, and k is 0.12 yr^{-1} (LandTech, 1994) for MSW containing both degradable and non-degradable components. K and Lo may differ for mixed wastes and individual components. De la Cruz and Barlaz (2010) conducted a laboratory scale study to estimate the waste component-specific landfill decay rates, using laboratory-scale decomposition data as, shown in Table 2-9. The Lo of food waste was found to be the highest (about 300.7 m^3/Dry Mg), and the decay rate, k was about 0.289 yr^{-1} (De la Cruz and Barlaz, 2010).

Table 2-9 Laboratory-scale decay rates, methane yields, and moisture contents for various MSW constituents (De la Cruz and Barlaz, 2010; Eleazer et al., 1997; and Tchobanoglous et al., 1993)

Components	**Average Decay Rate, k ($Year^{-1}$) (De la Cruz and Barlaz, 2010)**	**Methane Generation Potentail, Lo (m^3/dry Mg) (Eleazer et al, 1997)**	**Moisture Content (Wet wt %) (Tchobanoglous et al., 1993)**
Office Paper	3.08	217.3	6
Newspaper	3.45	74.3	6
Corrugated Containers	2.05	152.3	5
Coated Paper	12.6821	84.4	6
Food	15.02	300.7	70
Grass	31.13	144.4	60
Branches	1.56	62.6	30
Leaves	17.82	30.6	30

2.6 Properties of Municipal Solid Waste

Properties of MSW mainly depend on the characteristics of individual components present in the MSW mass. Because of the presence of various materials, MSW is extremely heterogeneous, making it very difficult to determine the individual properties. It is very important to know the engineering properties of MSW to monitor and evaluate the performance of the landfill. According to Manasslero et al., (1997) it is extremely difficult to determine the engineering properties due to the following reasons:

- Difficulties in sampling MSW which simulates the in situ condition,
- Lack of generally accepted sampling procedures for geotechnical characterization of waste materials,
- Variations in properties of municipal solid waste with time,
- Inadequate levels of training and education of the personnel on site for basic interpretation and understanding of the measurements, and
- Heterogeneity of the MSW within the landfill and its variations with geographical location.

Composition, moisture content, unit weight or density, hydraulic conductivity, porosity, compressibility, stiffness, and shear strength are usually the most important properties of any solid waste, and they determine the design of a landfill, particularly a bioreactor landfill. Several studies have been conducted to determine the engineering properties of MSW, but it is very difficult to apply the properties of waste from one site to another since the high heterogeneity differs spatially and temporally. Waste is more complicated than soil because of the potential for biological and chemical ongoing processes and interactions. It is a highly porous material, with solids and pore space distributed throughout the mass. The pore space may be filled with liquid and/or gas. This porous medium is often considered to be unsaturated soil (McDougall et al., 2004), most closely comparable to solid waste landfills in terms of structure, porosity and gas content. The solid phase also comprises a wide range of different material types with vastly different mechanical and physical properties, leading to the uncertainty and heterogeneity of the waste properties.

2.6.1 Waste Composition

MSW is the combination of various components such as paper, plastics, food scraps, rubber, leather, textile, wood and yard trimmings, metal, glass and other (fines). Characteristics of the individual components control the overall properties of waste mass. The percentage of waste within individual categories is important information for planning solid waste management programs, which includesevaluation of recycling programs, quantification of the degree of success in excluding banned items from the waste stream, quality of waste to be used as feedstock to an incinerator, quantification of organics to evaluate biogas possibilities, etc. Table 2-10 shows the typical range of percentages of the weight of MSW components in dry and saturated conditions, which plays a major role in the varying unit weights of MSW.

Table 2-10 Typical waste composition and unit weight (Landva and Clark, 1990)

Category	**Percent of total weight**	**Dry unit wt. (kN/m^3)**	**Saturated unit wt. (kN/m^3)**
Food waste	5-42	1.0	1.0
Garden refuse	4-20	0.3	0.6
Paper products	20-55	0.4	1.2
Plastic, rubber	2-15	1.1	1.1
Textiles	0-4	0.3	0.6
Wood	0.4-15	0.45	1.0
Metal products	6-15	6.0	6.0
Glass & ceramics	2-15	2.9	2.9
Ash, rock & dirt	0-15	1.8	2.0

Individual waste components have a wide range of moisture content, unit weight, and other engineering properties which can change with time, resulting in a significant percentage of waste particles behaving differently from soil particles due to their high compressibility (Dixon and Jones 2005). For example, organic MSW component has a high potential for loss of mass due to the degradation process. It is generally believed that degradation results in an increase in waste density, and hence unit weight (Dixon and Jones 2005).

2.6.2 Organic Content

Volatile solid (VS) content, or the organic matter content, is an indication of the degradation level of the waste mass and remaining gas generation potential of the waste. It decreases with decomposition, i.e. at deeper depths of landfill. MSW at an advanced stage of decomposition has low organic content; at surface level, where the MSW is in the initial stage of decomposition, it has more volatile solids (Barlaz, 1988). Organic content, also known as loss-on-ignition, is defined as the percent of weight loss on ignition at 550°C according to Standard Methods (APHA et al., 2005). Several studies have been conducted on volatile solid content in MSW mass, as shown in Table 2-11.

Table 2-11 Volatile solids of MSW reported in literature

Type of waste	**Volatile solids (%)**	**Reference**
Fresh degradable MSW	79	Barlaz et al. (1990)
Aged MSW	33	Gabr and Valero (1995)
Aged MSW	40.1-42.6	Mehta (2002)
Aged MSW	56	Gomes (2005)
Fresh MSW	91.5	Haque (2007)
Fresh MSW	76-84	Reddy (2009)
Fresh MSW	84	Al-Kaabi et al. (2009)
Fresh MSW	84	Sivanesan (2012)
Food waste	91.66-92.96	Zaman (2016)
Sorted degradable MSW	84.41-87.04	Sapkota (2017)

According to Barlaz et al. (1990), organic content of fresh MSW is about 79%. Haque (2007) estimated that the organic content of fresh MSW is about 91.5%. Reddy et al. (2009) reported that the organic content of fresh MSW collected from the Orchard Hills Landfill in Illinois ranges from 76 to 84%. According to Al-Kaabi et al. (2009) and Sivanesan (2012), organic content of fresh MSW is about 84%. Sapkota (2017) determined that the organic content in fresh and sorted biodegradable MSW is 84.41 - 87.04%. Zaman (2016) reported that food waste usually has higher organic content (91.66 - 92.96%) than other waste components.

Organic content of waste decreases with age and deeper depth. Gabr and Valero (1995) estimated the average organic content of aged waste from a depth of 19 m to be 33% percent. Mehta et al. (2002) determined the organic content of MSW from two test cells in Yolo County, California to evaluate the effect of leachate recirculation on waste degradation. The authors found the average organic content in the control (without leachate recirculation) samples and enhanced (with leachate recirculation) samples to be 40.1 and 42.6%, respectively. Gomes et al. (2005) showed the variation of organic content with depth ranging from 43 to 63% near the surface and 56% at 11 m depth at the San Tirso landfill in Portugal. Hossain and Haque (2009) showed that the organic content of MSW decreased from 94% in phase 1 of decomposition to 41% in phase 4 in the laboratory scale reactors.

2.6.3 Moisture Content and Field Capacity

It is important to consider the moisture content of municipal solid waste when estimating its heat or energy content, the amount of land required for the landfill, and transport requirements (Pichtel, 2005). Moisture content is very useful when the waste is being used as fuel. Moisture content varies according to the waste constituents and changes with time as the transfer of moisture takes place within the waste in the garbage can and truck (Vesilind et al., 2002). Moisture content of waste is commonly expressed as the percentage of water to the wet weight of the waste material. In soil mechanics, the water content is defined as the ratio of the mass of water to the mass of dry solids. It is normally denoted by “w” for soil in soil mechanics. Two types of moisture content are used for solid waste, dry weight and wet weight; however wet weight is more commonly used. The moisture content can also be expressed on a volumetric basis, where the volumetric water content is defined as the ratio of the volume of water to the total volume of air, solids, and water in the waste mass.

In the United States, the moisture content of MSW varies from 15 to 40% on wet basis, depending on composition, season of the year, and weather conditions (Pichtel, 2005). According to Gabr and Valero (1995), in 15 to 30 year old MSW landfills, moisture content varies from 30% at the surface to as high as 130% at greater depths. Xiang-rong et al. (2003) reported that moisture content gradually decreases from 60 to 20% with depth, with an average of 30% at the Tianziling Landfill. Geotechnical properties of fresh

MSW collected from the working face of the Orchard Hills Landfill were determined and reported by Reddy et al. (2009), and according to them, the dry gravimetric moisture content of MSW was 44% . Moisture content increases with increasing organic content and can be up to 120% for landfills across Canada (Landva and Clark, 1990). In a study by Gomes et al. (2005), moisture content of fresh waste and three-year-old waste were 61% and 117%, respectively. According to Taufiq, T. (2010), moisture content of fresh MSW from the working phase of the City of Denton Landfill in Denton, Texas, USA was an average of 37.5% by wet basis. Shihada et al. (2013) reported that the moisture content of fresh and landfilled degraded MSW were about 27.05% and 18.8 -31.15%, respectively. Moisture content varies, depending on the waste composition, organic content, geographic location, weather, and operating conditions. In Dhaka City of Bangladesh, the moisture content of fresh MSW ranges from 65 to 80% (Yousuf and Rahman, 2007) due to the presence of high organic content. According to Kumar et al. (2009), the moisture content of fresh MSW ranges from 17 to 65% and is dependent upon the population of the city.

It is important to know the moisture holding capacity or field capacity of waste to estimate the amount of moisture that needs to be added to a landfill before leachate generation and extraction can occur. When moisture added to the waste amounts to more than its field capacity, leachate is produced. Field capacity is the maximum moisture that can be retained by the waste mass without producing downward percolation (Beaven 2000). This amount of moisture, expressed as the percentage weight or volume of the total waste mass is denoted as the field capacity of MSW. Moisture retention is attributed primarily to the holding forces of surface tension and capillary pressure. Percolation occurs when the magnitude of the gravitational forces exceed the moisture-holding forces of the waste mass. The field capacity of MSW, on a volumetric basis, has been reported in literature as ranging from 14 to 44% (Table 2-12).

Table 2-12 Typical field capacity of MSW landfills

Field capacity (v/v)	Reference
29	Remson et al. (1968)
29-42	Holmes (1980)
30-40	Straub and Lynch (1982)
20-30	Korfiatis et al (1984), Owens et. al (1990)
14	Zeiss and Major (1993)
29	Schroeder et al. (1994)
44	Bengtsson et al. (1994)
31.5-36.9	Wu et al. (2012)
20.3-27.9	Breitmeyer and Benson (2011)

Increased moisture content enhances methanogenesis and stimulates microbial activity by providing better contact between insoluble substrates, soluble nutrients, and microorganisms (Barlaz et al., 1990) and is the major contributor to the formation of leachate. According to Christensen and Kjeldsen (1989), the production rate of landfill gas is proportional to the moisture content of the wastes and is between about 20% moisture and close-to-waste saturation. If moisture is less than 20% , the microbial activities cause a decrease in gas production. The concept of the bioreactor evolved from the realization that the addition of water enhances the decomposition process of waste. Understanding the moisture distribution within MSW in a landfill due to leachate recirculation is vital to the design and optimized operation of a leachate recirculation system. It is advantageous for a landfill operator to be able to measure the in situ moisture content of a landfill as it relates directly to the quantity and timing of leachate formation and biodegradation activities that affect landfill gas production (Yuen et al 2000).

2.6.4 Unit Weight

Unit weight of MSW is defined as the weight of waste per unit volume. It can vary significantly, based on the composition of the waste, the degree of compaction, the type of cover soil, and the stage of decomposition. Unit weight is a critical parameter of a landfill operation because it affects the permeability of the waste, and thus affects the moisture movement inside the waste, as well as gas production. Bulk density (ρwet) is similar to the unit weight and is defined as the total mass of the waste (solid + water) within a unit volume (Vt) of solid waste. The dry density (ρdry) is defined as the total mass of dry solids within a unit volume of solid waste.

$$\rho_{wet} = \frac{M_S+M_W}{V_t} \qquad (2.4)$$

$$\rho_{dry} = \frac{M_S}{V_t} \qquad (2.5)$$

Unit weight/density of MSW is an important parameter of landfill operations. To enhance waste decomposition and gas production, moisture inside the waste should be transported all around the solid waste mass. Permeability is directly related to the unit weight/density of the waste, so it's very critical. Density of MSW is also necessary for many engineering analyses of landfill systems, including slope stability, geomembrane puncture, pipe crushing, and landfill capacity evaluation. Compaction of wastes at a landfill is the main factor that controls short-term density and resulting placement efficiency of wastes in the landfills (Hansen et al. 2010). Maximizing waste density reduces landfill space requirements to prolong the life of a facility (Ham et al., 1979). Density influences the stability of a landfill. High densities are generally associated with high shear strengths and high frictional angles. Combined moisture-density characteristics influence the hydraulic response and compressibility of wastes. Overall, the moisture-density characteristics of MSW are critical for both the operation of landfills and the engineering response of wastes (Hanson et al., 2010).

Unit weight/density of MSW varies significantly between sites and within a single site due to heterogeneous material (Dixon and Jones 2005). A wide range of unit weight values of MSW have been found in several studies. Landva and Clark (1990) determined that unit weight of MSW ranges from 6.8 to 16.2 kN/m^3 in Canadian landfills by in-situ unit weight measurements. According to Fassett et al. (1994), unit weights range from 3 to 9 kN/m^3 for fresh waste with poor compaction, 5 to 7.8 kN/m^3 for moderate compaction, and 8.8 to10.5 kN/m^3 for good compaction. Fassett et al., (1994) proposed that other factors should be recorded along with measured unit weights, such as MSW composition, daily cover and moisture content, method and degree of compaction, the depth at which the unit weight is measured, and the age of the waste. Zekkos et al. (2005) determined that the values of in-situ MSW unit weight varied from 3 - 20 kN/m^3.

Some studies have been conducted on laboratory compaction of MSWs, in particular for fresh wastes. Common soil testing procedures, i.e. standard proctor tests, were commonly followed to estimate unit weight for fresh wastes in geotechnical investigations of waste characteristics. According to Harris (1979), the maximum dry unit weight of 7.1 kN/m^3 and 58 percent optimum moisture content was found from standard proctor tests on wastes obtained from landfills in England. Reddy et al., (2009 b) found 420 kg/m^3 maximum dry density and 70% optimum moisture content in fresh wastes obtained from a landfill in Illinois.

2.7 Factors Affecting Waste Degradation and Methane Production

Various factors affect the degradation of waste, as well as the quantity and rate of methane production. Waste composition, or the degradable organic content and smaller particle size, increase methane production to great extent. Environmental factors such as moisture content, nutrient content, temperature, pH, and amount of toxic substances have been seen to affect the waste degradation. Various technologies such as leachate recirculation and the addition of nutrients have been observed as the main agent for accelerating the biodegradation of waste by several studies (Barlaz et al., 1982; Barlaz et al., 1989, Christensen et al., 1992; El-Fadel et al., 1996; Wraith, 2003; and Wraith et al., 2005).

2.7.1 *Waste Composition*

The amount of methane produced by a waste mass depends on the organic content of the waste. Waste composition varies with geographical location and depends on economic conditions, lifestyles of the people, and the industrial structure. Was consists of several components such as food, paper, wood, textile, plastic, styrofoam, metal, glass, and fines, and they all have different amounts of organic contents and degrade at different rates. However, not all organic materials can to be degraded by the bacteria. With the decrease of the degradability substrate, the lignin content increases in the waste. Furthermore, different components of waste degrade at varying rates. Organic waste constitutes the highest fraction of total MSW mass and is higher in developing than in developed countries (Guermond et al., 2009), meaning that the potential for gas generation is higher in developing countries. In the organic waste mass, food waste has the highest potential of gas generation, as food is highly degradable due to presence of moisture.

2.7.2 Particle Size of Waste

Particle size of waste plays a vital role in the waste degradation process. MSW with a reduced particle size provides more homogeneity than unprocessed MSW. For example, shredding waste increases the rate of decomposition and methane production (Ham and Bookter, 1982), as the surface area of contacts between the key refuse constituents increases (Barlaz et al. 1990). In addition, waste shredding can increase oxygen utilization and the rate of decomposition, i.e. ultimately resulting in early methane production (Ham and Bookter, 1982; Otieno, 1989). A study by Buivid et al. (1981) showed the opposite proposition They reported that shredding waste to particle sizes in the range of 250-350 mm produced 32% more methane after 90 days than MSW with particle sizes of 100 – 150 mm; 100 to 150 mm particles produced 16 times more methane than a finely shredded MSW of less than 25 mm particle size. Their study showed that smaller particles produced less methane. This might be because the smaller particles increase the rate of hydrolysis and acid formation, which ultimately decreases the pH and postpones the methane production. A study conducted by Warith et al. (2005) reported that shredded MSW produces leachate with higher peak COD concentrations and slightly lower minimum pH levels than unprocessed MSW. Hence, particles that are too small might cause rapid waste hydrolysis and acid formation, which has a negative impact on methane production. Sponza et al. (2005) showed that the shredding of MSW has a positive impact on degradation in anaerobic bioreactors with leachate recirculation. Three types of reactors were compared: one with raw waste, one with shredded waste, and one with compacted waste. At the end of the experiments (after 57 days) the reactor with shredded waste had the lowest COD and VFA concentrations and the highest methane percentage.

2.7.3 Temperature

The landfill temperature is one of the major factors contributing to gas and leachate generation at landfill sites and subsequently affects the characteristics of LFG and landfill leachate (El-Fadel et al. 1997). The spatial distribution of temperature over time in a landfill located in Michigan (US) was studied by Yesiller et al. (2003), with the conclusion that the temperature of waste is significantly affected by seasonal variations;

placement, age, depth, location, and moisture content of the waste. Temperature affects most of the microbial activities and reaction kinetics, i.e. biodegradation of waste, and gas generation and emissions. So at a particular range of temperature, a landfill produces more methane due to increased microbial activities and reaction kinetics. Both anaerobic and aerobic degradations are exothermic reactions, so landfill temperature is expected to be higher than the atmospheric temperature. The amount of heat generated during anaerobic degradation is only 7% of that generated during aerobic degradation (Christensen and Kjeldsen 1989; Rees 1980; Bingemer and Crutzen 1987). Size and height of the landfill, climatic conditions, and landfilling operations affect the temperature of landfill by determining the circumstances in which microbial decomposition occurs (Wang et al. 2012). The impact of temperature on landfill gas emissions and landfill leachate is vital for the improvement of long-term landfill management techniques, which are significant for minimizing gas emissions, accelerating waste stabilization, and shortening the post closure time. During the biodegradation phase, in the transition from acidogenesis to methanogenesis phases, the quality of leachate varies significantly.

According to Blakey et al. (1997), temperature is an important factor affecting the methane content of landfill gas. Hartz et al. (1982) studied seven different temperatures, ranging from 21°C to 48°C, and observed 41°C as the optimum temperature for short-term methane production. Mata-Alvares and Martina Verdure (1986) stated that the optimum temperature for MSW degradation was 34°C to 38°C, which was independent of leachate recirculation. Rees (1980) identified that in a conventional anaerobic landfill, it is important to maintain the temperature at about 45°C. If landfills are operated at optimum temperatures, the rate of gas production and refuse stabilization are increased. In warmer climates, the transition from the acetogenic to methanogenic phase of the landfills can be shortened significantly. According to Robinson (2007), the transition period from the acetogenic to the methanogenic phase of the landfills in countries with warmer climates is two or three times less than the countries with moderate temperatures. A rapid transition from acidogenesis to methanogenesis can reduce the content of VFAs (volatile fatty acids) in leachate, rendering low BOD and BOD/COD ratios and ultimately producing more methane.

Christensen and Kjeldsen (1989) performed laboratory experiments and found that the methane production rate increases when the temperature is raised from 20 °C to 30 °C and 40 °C, but higher temperatures might not be useful for microbial activity in the waste. In a study performed by Tchobanoglous et al. (1993), a significant reduction in methane generation was observed with temperatures less than 20 °C and greater than 70 °C. In another study, the effect of temperature was seen on waste degradation in laboratory scale landfill reactors. Three temperatures were chosen: 25 °C, 37 °C, and 60 °C, and 37 °C was found to be the most favorable temperature for enhanced methane generation (Buivid et al., 1981), as mesophilic bacterial is active in temperatures from 30 – 40 °C. Higher temperatures increase microbial activity, with activity roughly doubling for every 10 °C increase within the optimal range (Khanal, 2008) up to 60 °C, as thermophilic bacteria is active at temperatures from 50 – 60 °C.

2.7.4 pH Level

The chemical and biological process of waste decomposition is significantly influenced by the pH of the recirculated leachate. According to Valencia et al. (2009), pH is possibly the ‘driving force’ to trigger all processes. Anaerobic degradation of waste involves decomposition by bacteria to produce up to about 70% of methane (NAS, 1977; Gujer and Zehnder, 1983) with the presence of water and contact with molecular oxygen eliminated. During the initial stages of anaerobic decomposition, organic acids form, resulting in an acidic pH. As these organics begin originate, the pH rises, as the acids are converted to methane. Different bacteria are active in various phases of degradation. The bacteria active in the acidogenic phase, i.e. acidogens, prefer a pH of 5.5-6.5. Methanogenic bacteria or methanogens are responsible for methane production and are active when the pH is 7.8-8.2 (Khanal, 2008). When acidogens and methanogens reach equilibrium, during the process of removing organic acid, the pH naturally stabilizes at around 7 (Metcalf and Eddy, 2004). Some other studies have shown that a pH range of 6.4 - 7.2 is favorable for methanogens (Chugh et al., 1998; Yuen et al., 2001). According to Warith (2003), the ideal methanogenic bacterial activity occurs in environmental conditions within a pH range of 6.8 to 8.0. If the pH drops below 6.6, methanogens are significantly inhibited, while pH below 6.2 is toxic (Metcalf and Eddy, 2004). According to Christensen et al., (1996), any drop in the pH value below 6.8 slows down the microbial activity and growth of methanogenic microorganisms. If the methanogenic activity is inhibited by other factors, such as oxygen and hydrogen, etc. in a well-established methanogenic media, the conversion of acetic acid to methane and carbon dioxide decreases. This leads to an accumulation of the acids, thereby decreasing the pH, and may stop the generation of methane (Christensen et al., 1996). Within the optimum pH range, methanogens grow at high rate and result in maximum methane production. The rate of methane production is seriously limited when the pH level is lower than 6 or higher than 8 (Barlaz et al, 1987).

2.7.5 Moisture Content and Leachate Recirculation

Moisture content is one of the most important factors of waste degradation, as it plays an important role in microbial activities. As methane in the landfill is produced from the anaerobic degradation process of waste with water, moisture is the single most important factor of methane production (NAS, 1977; Gujer and Zehnder, 1983). It acts as a medium to transfer the bacteria and nutrients which increases gas production and waste decomposition in the landfill. If the moisture content of the waste in landfill exceeds its field capacity, the free moving liquid carries the bacteria and nutrient to other areas within the landfill, providing favorable environment for gas production. The bioreactor landfill technology relies on maintaining the optimal moisture content near field capacity (approximately 35% to 65%) by recirculation of leachate and adds liquids when it is necessary to maintain that percentage (US EPA). According to several studies, methane generation rate increases with an increase in moisture content (Barlaz et al., 1990; Mehta et al., 2002; Rees, 1980; Christensen et. al., 1996; Warith et al., 2005), as water is the key factor in accelerating the biochemical decomposition of organic substances (Klink and Ham, 1982). When the moisture content in a landfill increases, it limits the oxygen transport from the atmosphere, providing favorable conditions for the anaerobic process. In addition, increasing the moisture content helps to facilitate the exchange of substrate, nutrients, buffer, and dilution of inhibitors; and distributes the micro-organisms inside the landfill (Christensen et. al., 1996; Warith et al., 2005). A study by Rees (1980) showed that with the increase of water content from 25 to 60%, the percentage of methane and gas produced also increased, as shown in Figure 2-11. Rees (1980) also studied the effect of moisture circulation on wastes and found that maximum methane content is generated by daily circulation of water (Figure 2-12). Another study which showed a similar result was conducted on two cells, one with and one without controlled moisture addition. Again, the rate of gas generation increased with an increase in moisture content (Mehta et al., 2002). Faruquhar and Rovers (1973) observed that the maximum amount of methane was generated at moisture contents of 60 to 80% on wet weight basis. The effect of various moisture contents on the methane generation rate was also studied by Hernandez-Berriel et al. (2010) in laboratory scale bioreactors. They found that the methane generation rate increased as the moisture content increased from 50 to 70%, with 70% moisture content being optimum for methane production. The methane generation rate was significantly inhibited at 80% moisture content due to washout of nutrients.

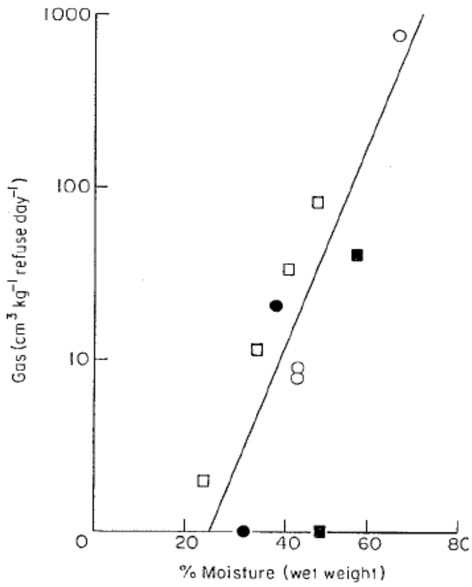


Figure 2-11 Plots of moisture content vs. methane generation rate by Rees (1980)

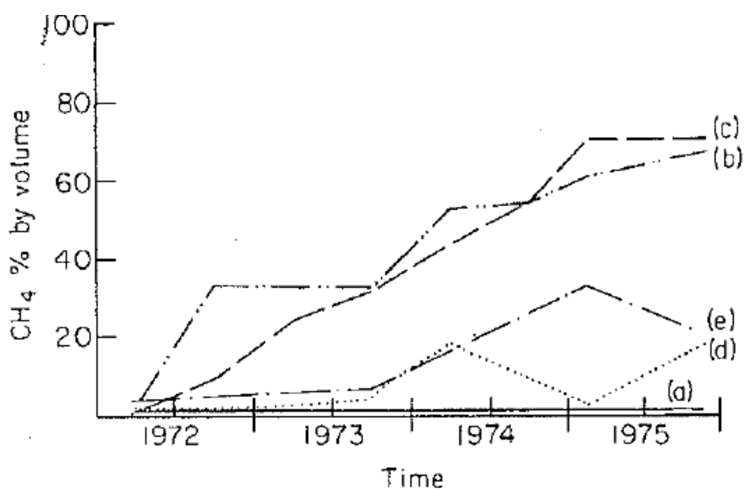


Figure 2-12 Effects of water content on the methane content of landfill gas (a) Dry waste; (b), (c) Daily liquid application; (d),(e) Initially saturated (Rees, 1980)

Leachate recirculation is one of the most effective techniques to increase moisture content inside a landfill. It helps to distribute moisture, nutrients and enzymes inside the waste; neutralize the pH; dilute toxic compounds; recycle and distribute methanogens (Reinhart and Al-Yousfi, 1996); and spread micro-organisms inside the landfill (Christensen and Quail, 1996; Warith et al., 2005). According to Pohland (1975), landfill with leachate recirculation can be stabilized within two or three years, while conventional landfills require several decades to be stabilized. Thus, a bioreactor landfill helps to minimize adverse environmental impacts. Bioreactor landfill technology maintains the optimal moisture content of waste near field capacity (approximately 35 to 65%) and adds or removes liquids when it is necessary to maintain that percentage (US EPA).

2.7.6 Nutrient Content

Various nutrients, such as carbon (C), nitrogen (N_2), phosphorus (P), sodium (Na), potassium (K), and other trace materials are required for the microorganisms in the landfill to be active. Trace metals that enhance methane production are iron (Fe), cobalt (Co), molybdenum (Mo), selenium (Se), calcium (Ca), magnesium (Mg), zinc sulfide (ZnS), copper (Cu), manganese (Mn), tungsten (W), and boron (B). Methanogens require nitrogen (N_2) and phosphorus (P) for their growth. All necessary nutrients and traces of heavy metals are available in most landfills, but heterogeneous characteristics of waste may limit the availability of nutrients inside a landfill (Rees, 1980; Christensen et al., 1996). High concentrations of heavy metals may slow bacterial growth and consequently decrease gas production. If greater amounts of digested nutrients are available, gas generation will increase.

2.7.7 Concentration of Oxygen and Hydrogen

The anaerobic process is considered sensitive to inhibitors. The activities of methanogenic bacteria are hindered significantly and sometimes stopped by the presence of oxygen (O_2). In the initial phase of decomposition, the oxygen that diffuses from the atmosphere into the landfill is consumed by aerobic bacteria in the top layers of the landfill. Under normal conditions, aerobic bacteria allows the solid waste to readily consume the oxygen and limit the aerobic zone of the compacted waste (Warith, 2003), then the anaerobic phase starts with very low or no concentration of oxygen. In the later phase, there may be oxygen intrusion in the landfill due to operational activities. During the pumping of landfill gas, extensive pumping may create a substantial vacuum in the landfill, forcing air to fill it. This will extend the aerobic zone in the landfill refuse and eventually prevent the formation of methane in these layers (Christensen et al, 1989).

Hydrogen is another inhibitor to methanogenic bacteria. The fermentative and acidogenic bacteria produce hydrogen, whereas the methanogenic bacteria use the hydrogen as a substrate to produce methane. During the acidogenic phase, propionic acid and butyric acids are produced. The conversion of propionic acid requires a very low hydrogen pressure of less than $9X10^{-5}$ atmospheres (Christensen et al., 1989). There may be an increase in the partial pressure of hydrogen, which causes the generation of propionic and butyric acids with no further conversion. For example, volatile organic acids are produced and accumulated, which reduce the pH and inhibit the methanogenic bacteria (Christensen et al., 1989).

2.7.8 Concentration of Toxic Substances

In addition to oxygen (O_2), hydrogen (H_2), acidic pH, and high concentrations of heavy metals, a number of other compounds can inhibit the biodegradation of solid waste. These inhibitors are carbon dioxide (CO_2), sulfate (SO_{4-2}), and high concentrations of cations such as sodium, magnesium, ammonium, and specific organic compounds. Cations have been observed to stimulate anaerobic decomposition at low concentrations, while inhibiting it at high concentrations (Christensen and Kjeldsen, 1989). The CO_2 may increase the redox potential (Hansson, 1982), which has an effect on the conversion of acetic acid to methane (Christensen et al., 1996) or raise the impairment of the methanogen cell membrane function by increasing its fluidity through CO_2 dissolving in the cell membranes of methanogens (Senior and Kasali,1990).

2.7.9 Inoculum Addition

Many studies suggest adding inoculum to the MSW and leachate to enhance waste decomposition. Various kinds of animal manure, municipal sewage sludge, septic tank sludge, and old MSW have been recommended as inocula by many researchers. (This section is described elaborately in section 2.3.2.) The effect of adding sludge adding on the MSW biodegradation has been noted in a number of studies by Leuschner (1982), Pacey (1989), and Warith (2002). According to their studies, the addition of sewage sludge has both positive and negative effects on the MSW biodegradation and methane generation. The positive effects can be observed when the landfill environment is optimum (pH neutral) for methanogenic bacteria and the bacteria have been established (Christensen et al., 1992). In this state, the landfill is a source of nutrients and active methanogenic bacteria, which increases the moisture content. On the contrary, according to Barlaz et al. (1990), the addition of sludge to fresh waste can cause an accumulation of acid, which decreases the pH and inhibits the methanogenic bacteria. Erses and Onay (2003) studied the utilization of external leachate recycled from old landfills and found that desired acclimated anaerobic microorganisms, low organic content, and a higher buffer capacity in a young landfill could be a promising leachate management strategy for faster waste stabilization.

2.7.10 Lift Design, Daily Cover and Compaction of Waste

List thickness, daily cover, and compaction of waste also enhance waste biodegradation in the landfill. The lift thickness has a negative effect on the waste decomposition. According to Ham et al. (1982), a cell with a 2 m deep lift produced higher leachate concentrations and took longer to stabilize than a cell with a 1.2 m deep lift. Doubling the lift depth from 1.2 to 2.4 m also doubled the concentration of leachate and stabilization time. During compaction of landfill layers, the first layer should not be compacted so that readily degradable organics can decompose aerobically and stabilize before the addition of subsequent lifts (Stegmann, 1983). Reinhart et al. (2002) concluded that increased MSW compaction reduces waste's ability to move moisture and enables the waste to achieve a level of saturation with less moisture addition because waste hydraulic conductivity is inversely related to waste density. Reinhart and Townsend (1998) suggested that the use of high permeability soils and/or alternative daily covers may reduce ponding and horizontal movement of leachate. Alternative daily cover materials include mulched or composted yard waste, foam, carpet, clay/cellulose additives, and geotextiles. The use of these alternative materials may result in savings of landfill space and costs, increase of waste hydraulic conductivity within the landfill, and extended life of the leachate drainage layers (Wiles and Hare 1997).

2.7.11 Pre-treatment

The decomposition process can be enhanced with the pre-treatment of MSW. Pre-treatment enhances the acidogenic stage and decreases the accumulation of organic acids. This method is based on the stabilization of part of the waste through aerobic processes which dilute the organic acids and cause a balance between the acidic phase and the methanogenic bacteria (Ham et al., 1982; Stegmann, 1983; Beker, 1987).

Placing fresh waste on top of the composted waste layer caused a shorter acidogenic stage and enhanced the methanogenic stage in the reported study.

2.8 Summary

Manures can be great additives for achieving faster decomposition during the early stages of landfilling (Yazdani, 2010). At later stages, the MnP enzyme can be used to break down the lignin content in waste (Jayasinghe, 2013). The success of the using enzymes for leachate augmentation has been proven to be effective in lab scale studies, but not in field scale. No study has been done on leachate augmentation by enzymes using fresh waste.

Chapter 3

Methodology

3.1 Introduction

This chapter includes the methodology to accomplish the research objectives outlined in Chapter 1, including the procedures followed for a laboratory-scale study of a biocell and the key features and components of a field-scale biocell. Laboratory tests and experiments were conducted to analyze the effects of additives on degrading organic fractions of MSW and food waste relative to enhancing gas production. Different laboratory tests such as physical characterization, moisture content, volatile organic content test, pH, BOD, and COD tests were conducted, along with the measurements of rate, volume, and composition of generated gas. It also includes the monitoring and data collection plan related to leachate redistribution within the field biocell.

3.2 Study Plan

The current study is divided into two major experimental programs: a laboratory-scale study and field application of a sustainable landfill biocell. A workflow diagram of this study is shown in Figure 3-1. The study started with a laboratory-scale biocell simulation, with reactors to evaluate the effects of nutrients such as manure and sludge on methane production from organic fraction of municipal solid waste (MSW) and food waste. Two steel waste containers with dimensions of 21 ft. by 8 ft. by 8 ft. were custom-designed to be used as biocell and control cells in the field. Based on the results from the experimental study, the biocell and control cells were installed in the field and were operated anaerobically for almost 14 months. Critical parameters of the landfill operation, such as landfill gas and leachate, were monitored periodically. Performance monitoring and evaluation of the biocell were carried out based on the experimental results from the laboratory simulation and field application of the biocell.

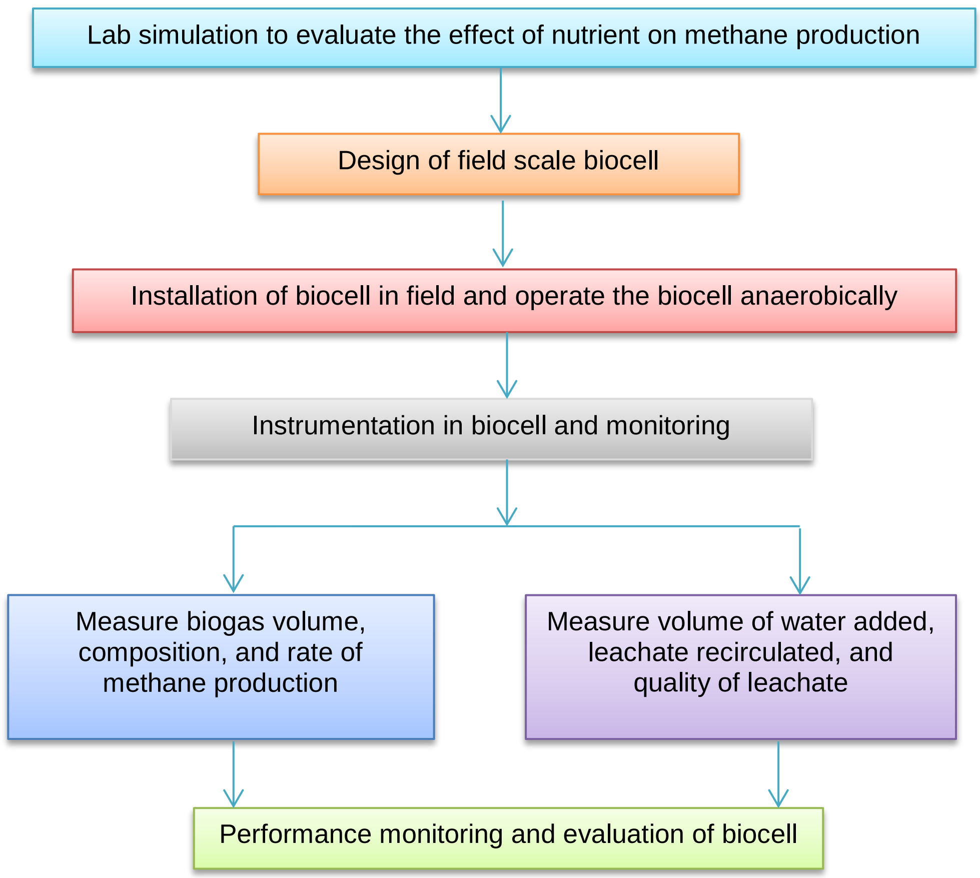


Figure 3-1 Flow chart of experimental program

3.3 Concept of Biocell

Traditional landfilling is not a long-term sustainable solution and has negative impacts on the environment and urban sustainability. Therefore, the proposed concept of a perpetual landfill or sustainable waste/material/resource management is the future of solid waste management practice. If certain measures are added, such as up-front removal of plastics, glass, and metals (recyclables), sanitary landfills can be transformed into sustainable/perpetual landfills that can be used in one location in perpetuity to generate renewable energy as organic waste completely degrades. A material recovery facility (MRF), as part of the landfill facility, can process the non-biodegradable plastics, glass, metals, and inorganics. However, with successful source separation of recyclables, this step can be avoided and all collected mixed waste can go to landfill. Degradation of landfilled organic wastes is faster than the degradation of solid waste mixed with non-degradable components. Moreover, adding nutrients such as enzymes, sludge, manure, etc. to the waste feedstock in a biocell accelerates the degradation process so that it occurs even faster than in a bioreactor, resulting in more biogas being produced and faster recovery of landfill space (Jayasinghe et al, 2011; Alkaabi et al., 2009; Yazdani, 2010). An active landfill can continue indefinitely for a perpetual landfill (biocell) under a sustainable material management system. The cells serve as temporary facilities, with biogas recovered for power generation. Since non-degradable plastics, glass, and metals are removed up-front, all waste in landfill cells is degradable.

3.4 Experimental Study on Biocell

The laboratory-scale simulation study on biocell is described in subsequent sections.

3.4.1 MSW and Food Waste Collection

Six bags of municipal solid waste were collected from random locations of the working face of The City of Denton landfill in March, (2016). The collected bags were tagged chronologically from one to six, and based on a study conducted by Taufig (2010), each contained approximately 25 – 30 lbs. of manually collected MSW samples.

Food waste was collected from Walmart in Denton, Texas; the City of Denton landfill; (Figure 3-2b) and the University Center (UC) dining hall at the University of Texas at Arlington (UTA) in Arlington, Texas. Approximately, 15 to 20 pounds of waste from fruits and vegetables were collected from Walmart in Denton, in two five-gallon buckets. Approximately 60 pounds of waste from meat, seafood, and grain products (rice, bread etc.) were collected from the UC dining hall.

Collected samples were taken to the Civil Engineering Laboratory Building (CELB) at UTA in plastic bags and were kept inside the environmental growth chamber (cold room) at 4 ^{o}C (38 ^{o}F) for preservation of moisture and other initial properties of waste which are shown in Figure 3-3.

(a) (b)

Figure 3-2 Sample collection from the (a) working face of the City of Denton Landfill; (b) Walmart, Denton

(a) (b)

Figure 3-3 a) Stored sample in cold room; b) Environmental Growth Chamber (Cold room and hot room)

3.4.2 Inoculum Collection

Cow, pig, and horse manure were used for this experiment because the high nitrogen ratio, low acidic bacteria, and increased hydraulic retention time of aged manure made it ideal for waste decomposition (Yazdani, 2010). A 20-pound bag of 9 -12 month old cow manure was obtained from Calloway Nursery in Dallas; 9 - 18 month old pig manure was obtained from the Colvin Creek Farm/Maypiggen, and 1 week old horse manure was obtained from the Triple H farm. Older horse manure was difficult to find in local nurseries. The samples were stored at room temperature. Two 5-gallon buckets of sludge were collected from the City of Denton Landfill as a source of microbes and to act as a buffer to neutralize the acidic environment inside the reactor. To maintain the anaerobic conditions of the sludge before adding it to the reactors, the samples were stored in an air tight container. Due to the potentially high redox value of manganese peroxide (MnP), it is described as a true lignin degrader (Martinez et al., 2005). It is soluble in water. The Santa Cruz Biotechnology Company, which supplies products for biomedical research, supplied the MnP enzyme used for this experiment and shipped it in blue ice since it is recommended that it be stored at -20 ^{o}C. The product number (EC number) of the enzyme used for this experiment is 1.11.1.13. The MnP enzyme was in the form of frozen, dried, brown-colored, amorphous powder. It was stored in a freezer at -20 ^{o}C and placed between two blue ice packs. The 30% hydrogen peroxide (H_2O_2) with product number 14411, obtained from the Hach Company, was used to activate the MnP enzyme.

3.4.3 Waste and Inoculum Combination

The physical composition of the MSW samples was determined by wet weight basis. The MSW was sorted manually and categorized into paper, plastics, textiles, food waste, Styrofoam and sponge waste, metals, glass, yard and wood waste, construction debris, and others. The percentage of paper, food waste, textiles and leather, and yard and wood waste was fixed according to the physical composition of degradable waste. Therefore 50% paper, 20% food waste, 15% textiles, and 15% yard and wood waste were selected for the reactors. Simulated biocell landfill reactors were built in the laboratory to analyze the gas generation and the effect of enzymes and manure on the degradation of the organic fraction of MSW and food waste. Five pairs of MSW reactors and four pairs of food waste reactors were built as landfill biocell simulators. Each MSW reactor was filled with 2 lbs. of municipal solid waste and 4 lbs. of food waste with varying inoculum types, as shown in Table 3-1. Five pairs of MSW reactors (M1 to M10) were assembled with organic fractions of MSW, and four pairs of food waste reactors (F1 to F8) were fabricated with food waste. Ten (10) percent sludge was added to all of the reactors. In the control reactors (M1, M2, F1 and F2), only organic fractions of MSW and food waste were used. Six percent of cow manure, pig manure, and horse manure were mixed in three pairs of MSW reactors (M3 to M8) and three pairs of food waste (F3 to F8). The manganese peroxide (MnP) enzyme was used in one pair of MSW reactors (M9 and M10).

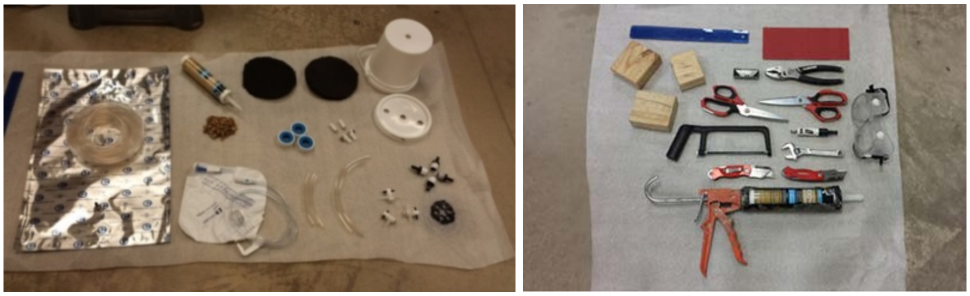
Table 3-1 Combinations of feedstock and inoculum for labrotary experiment

Reactors	**Waste Type**	**Sludge**	**Manure and Enzyme**
M1, M2	Organic MSW, 90%	10%	-
M3, M4	Organic MSW, 84%	10%	Cow Manure, 6%
M5, M6	Organic MSW, 84%	10%	Pig Manure, 6%
M7, M8	Organic MSW, 84%	10%	Horse Manure, 6%
M9, M10	Organic MSW, 84%	10%	MnP Enzyme, 0.00000213%
F1, F2	Food Waste, 90%	10%	-
F3, F4	Food Waste, 84%	10%	Cow Manure, 6%
F5, F6	Food Waste, 84%	10%	Pig Manure, 6%
F7, F8	Food Waste, 84%	10%	Horse Manure, 6%

3.4.4 Reactor Setup

The experiment was conducted in twenty reactors incubated under laboratory conditions at mesophilic temperature of 37 ^{o}C to simulate the actual landfill condition. The reactors were one-gallon HDPE wide-mouth plastic buckets (United States Plastic Corporation, OH) modified for gas and leachate collection, and liquid addition and recirculation. Different sizes of tubing, connectors, 22-liter Cali-5-BondTM gas bags, 2-liter leachate bags, valves, silicon sealants, washers, clamps, geocomposites, and gravels were required for building the reactors. Figure 3-4 shows the materials and equipment used to set up the reactors. Before filling them with waste, all of the reactors were checked for possible leakage, and leak tests were conducted, using a water head column, after the reactors were properly sealed. The reactors were monitored for one or two days to verify that there would be no significant leakage. The head difference at 12 and 48 hours was recorded to confirm that it was within permissible limits of 0.5 in. and 3 in. of water column, respectively (Mohammad Adil Haque 2007).

The bucket was modified, with 3 holes drilled on the lid and one hole on the bottom for gas and leachate collection and a threaded hose was used to connect tubes to the holes. The connections were made air tight, using thread tape. The two-way valves were connected to the tubes for leachate collection, and the three-way valve was connected for gas collection. These connections were made air tight with silicon sealant, which was left to dry for 24 hours. After filling the reactors, geocomposite layers were added to the top and bottom to stimulate the landfill liner system. To ensure better drainage of leachate from the bottom of the reactor, a pea gravel layer was used below the waste layer.



(a)

(b)

Figure 3-4 (a) Materials and equipment used for reactor building; (b) Reactor building process

All of the valves and sealed connections were tested for leaks by conducting a leak test. The pressure head difference method was used to test the connections. Initially the reactors were filled with water from a water tank and preserved a hydraulic head. The hydraulic head level was noted and the reactor was connected to a transparent tube and was observed for 48 hours. In the case of no change in the hydraulic head level, the reactor was considered to have passed the leak test. All of the reactors in the experiment passed the leak test. Paper, food waste, textiles and leather, and yard and wood waste were separated from the collected MSW. The MSW samples were cut, using scissors, into squares that were 1.5 inches by 1.5 inches before adding them into the reactors since shredding improves waste decomposition. The recommended size for particles for maximum gas production is one-fourth to one-fifth the diameter of the bucket. Sufficient moisture for microbial activities was provided by spraying water on each layer of filling. Proper compaction was also maintained throughout the process. Figure 3-5 shows the waste filling procedures. The reactors were filled 1 to 1.5 inches below the top level of the bucket to provide sufficient passage for gas to escape through gas collection outlet. The lids of the reactors filled with waste were then sealed, using a double layer of sealant. After sealing, the whole reactor setup was kept in the environmental growth chamber at a temperature of 37 °C. Figure 3-6 shows the schematic diagram for the reactor setup and reactors inside the environmental growth chamber.

Different combinations of sludge and manure were fed to the reactors that contained two pounds of organic MSW and four pounds of food waste. The addition of sludge and manure would provide microorganisms and sufficient nutrients to enhance the degradation process, as well as to neutralize the acidic environment by acting as a buffer. Out of the nine pairs of reactors built for this study, one pair had MSW with 10% sludge and one pair had food waste with sludge, which acted as control reactors. Feedstock of MSW with 10% sludge and 6% of three types of manure (cow manure, pig manure and horse manure) was prepared for three pairs of reactors. One pair had MSW with 10% sludge and MnP enzymes. Food waste with 10% sludge and 6% of three types of manure (cow manure, pig manure and horse manure) was prepared for another three pairs of reactors. For food waste, MnP was not tested, as it is a lignin degrading enzyme and food waste is mostly composed of cellulose and hemicellulose. The remaining reactor had feedstock of MSW with 10% sludge and 0.00000214 percent of MnP (manganese peroxidase) mixed with an organic fraction of municipal solid waste.

(a) (b) (c) (d) (e) (f) (g)

Figure 3-5 (a) Separating and shredding of waste; (b) Separated and shredded waste; (c) Mixing of waste and inoculum; (d) Filling of waste; (e) MSW reactor; (f) Food waste reactor; and (g) Reactor sealing

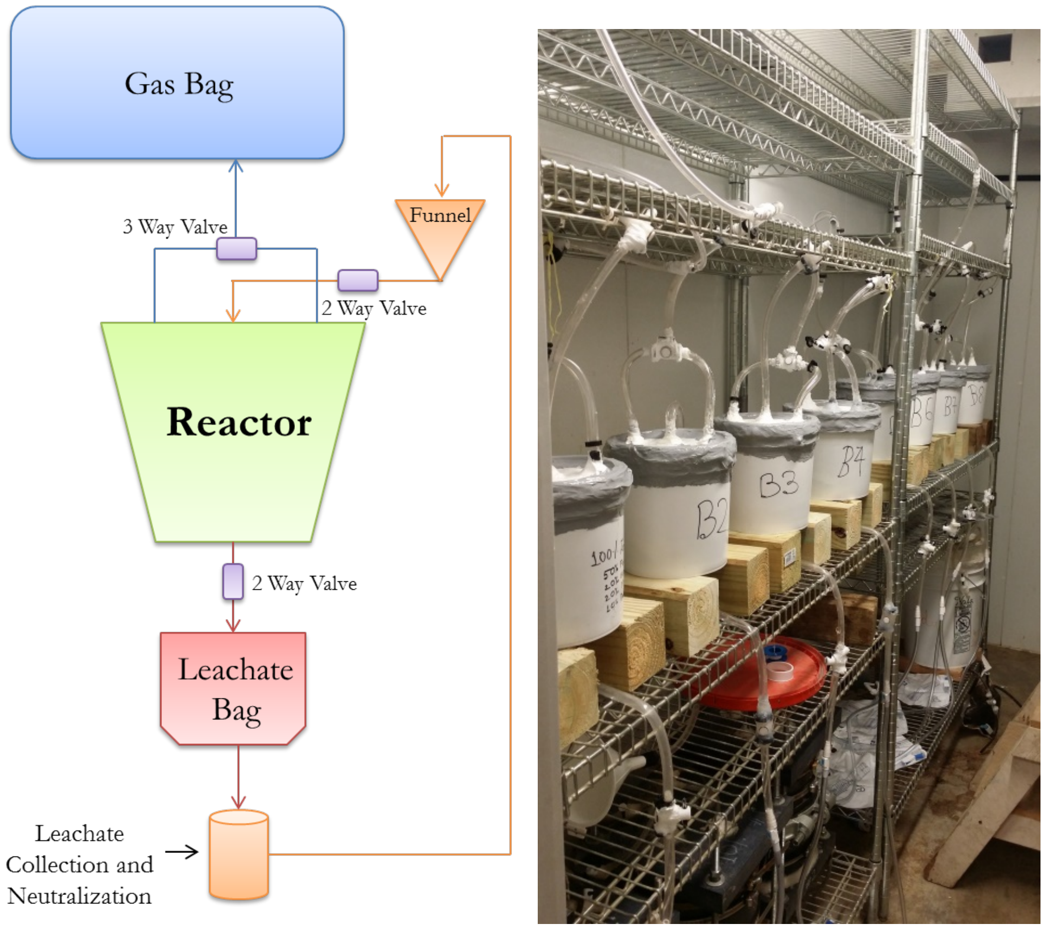


Figure 3-6 Laboratory scale landfill reactor setup

3.4.5 Reactors Operation and Monitoring

Routine monitoring was executed on the leachate and gas generation of the stimulated lab scale bioreactors. The volume, pH, COD, and BOD of generated leachate, along with leachate recirculation, were measured during the entire monitoring period. The gas monitoring program involved measurements of its composition, rate, and volume. The differences in moisture content and volatile solids before and after the degradation were also determined. The following sections discuss these activities.

3.4.5.1 Physical Properties of Waste and Inoculum

Physical properties such as waste composition, moisture content, and volatile solids of collected fresh MSW and food waste were measured before and after the laboratory experiment. Details of each follow.

Waste Composition: To determine the physical composition of the collected MSW and food waste, the bags of samples were emptied on large plastic sheets and were manually sorted into the categories of paper, plastic, food waste, leather and textile, wood and yard waste, metals, glass, styrofoam and sponge, construction debris, and others (fines). Any of the MSW that could not be placed under one of the mentioned categories, such as soil, lumps of mud, and objects too small to separate, was categorized as "others". The sorted components were weighed individually and were presented as a percentage of the total weight of waste in a bag. The MSW composition was then divided into two categories: degradable and non-degradable. Paper, food waste, leather and textile, and wood and yard waste were considered degradable components, and the remaining six categories were considered non-degradable.

Moisture Content: The moisture content was determined for the MSW and food waste in the collected bags in the initial phase of the experiment. In this study, the moisture content before and after degradation is referred to as the initial and final moisture content, respectively. Procedure 2540B in Standard Methods (APHA et al., 2005) was used to measure the moisture content of collected MSW samples. Wet, non-shredded MSW weighing approximately two pounds was dried in an aluminum pan for 24 hours at 105 °C, as shown in Figure 3-7. Food waste samples were dried for 5 to 7 days until a constant weight was achieved at 65 °C (±5°C) in the oven and measured for moisture loss. The percentage by weight of both wet and dry weight of refuse samples is expressed as the amount of moisture.

Figure 3-7 Determination of moisture content by drying sample in the oven

Equations 3.1 and 3.2 were used to determine the moisture content on a wet weight basis (W_w) and dry weight basis (W_d), respectively.

$$W_w = \frac{Mw}{Mt} \times 100\% \quad (3.1)$$

$$W_d = \frac{Mw}{Ms} \times 100\% \quad (3.2)$$

Where Mw is the mass of water, Mt is the total wet mass and Ms is the dry mass of water after drying.

Volatile solids: Organic content, also known as volatile solids (VS) and loss-on-ignition, is one of the main indicators of the degree of decomposition of MSW or food waste. Organic content in the waste decreases with decomposition. In this study, Method 2540-E (APHA et al., 2005) was used to measure the volatile solids of MSW and food waste. A sample of dry-milled refuse weighing approximately 50 grams was placed on a porcelain disk in a muffle furnace at 550 °C for 2 hours, until a constant weight was achieved. Figure 3-8 shows the residue or ash content of the MSW after the ignition. The percent of weight lost on ignition is the volatile organic content. Equation 3.3 was used to determine the percentage of volatile solids.

$$\text{VS (\%)} = \frac{Wl}{Wt} \times 100\% \quad\text{...(3.3)}$$

Where, W_l is the weight loss after burning and W_t is the dry weight of sample before burning.

Figure 3-8 Residue or ash content of MSW after the ignition

3.4.5.2 Gas Characteristics

Generated gas was collected in five-layer bags, and its volume and composition was measured on a regular basis. The Landtec GEM 2000 was used to measure the concentrations of methane (CH_4), carbon dioxide (CO_2), and oxygen (O_2) in the gas bags (Figure 3-9). An air-sampling pump (Universal XR Pump model 44XR) and Defender 330 scale were used to measure the volume of collected gas. The fixed rate of the flow of gas was recorded, then a stopwatch was used to record the time taken to empty the gas bags. The volume measurement process is shown in Figure 3-10.

Figure 3-9 Gas composition determination by Landtec GEM 2000

Figure 3-10 Gas sampling with Universal Sampler and Defender 330

3.4.5.3 Leachate Characteristics

Regular tests for pH, COD and BOD were conducted on generated leachate to monitor its characteristics and volume. Details are given in the following section.

pH: A benchtop Oakton pH meter, calibrated with the three-point calibration method, was used to measure the pH value of the collected leachate. A pH buffer of 4.00± 0.01, 7.00±0.01 and 10.00±0.01 was employed.

Chemical Oxygen Demand (COD): A spectrophotometer (Spectronic 200+) was used to measure the chemical oxygen demand (COD) of leachate samples. The spectrophotometer measured the absorbance of light and displayed the value on the screen for each sample. For each reactors, two tests were performed by diluting the leachate of MSW reactors and food waste reactors in 1:100 ratio and 1:200 ratio respectively. The dilution ratio needs to be fixed before the leachate undergoes COD and BOD tests. Therefore, to fix the dilution, COD tests were performed with different dilution factor and the dilution factor that fell below the calibration curve was used. One part of leachate was added to 99 parts of distilled water, as per ratio 1:100; 2.5 ml of diluted leachate was added to the COD vials. The vials were placed for 2 hours in a digester at a temperature of 150 °C. After being cooled to room temperature, the samples were placed in the spectrophotometer to determine the absorbance values. The COD values were obtained from the absorption percentage, using a calibration curve. A potassium hydrogen phthalate (KHP) solution with known COD values was used to generate a calibration curve. Figure 3-11 shows the calibration curve used for this study. COD values for corresponding absorbance values were determined, using the curve and adjusted according to the dilution factor to get the actual COD value for the leachate samples.

Biochemical Oxygen Demand (BOD): The amount of initial and final dissolved oxygen was measured to determine the biochemical oxygen demand (BOD). The HACH HQ 440d benchtop dissolved-oxygen measuring instrument was used to measure the dissolved oxygen. The BOD test was performed as per Standard 8043. The dilution factor of 1:100 was used (same as COD), and 300 ml BOD bottles were filled up to 75 percent with distilled water. The dilution water was prepared by adding a buffer pillow (one pillow for four liters of deionized water) and aerating it for two hours. A one-seed capsule was added to 500 ml of dilution water to make 3 ml of seed solution, which was also added to the BOD bottles. Then, the leachate samples were added. Each test was conducted three times by varying the volume of leachate added. The initial dissolved oxygen (DO) for each sample was measured using the calibrated BOD probe. The final dissolved oxygen was determined after keeping the samples at a temperature of 20 °C for 5 days. The five-day BOD_5 was calculated for all of the samples.

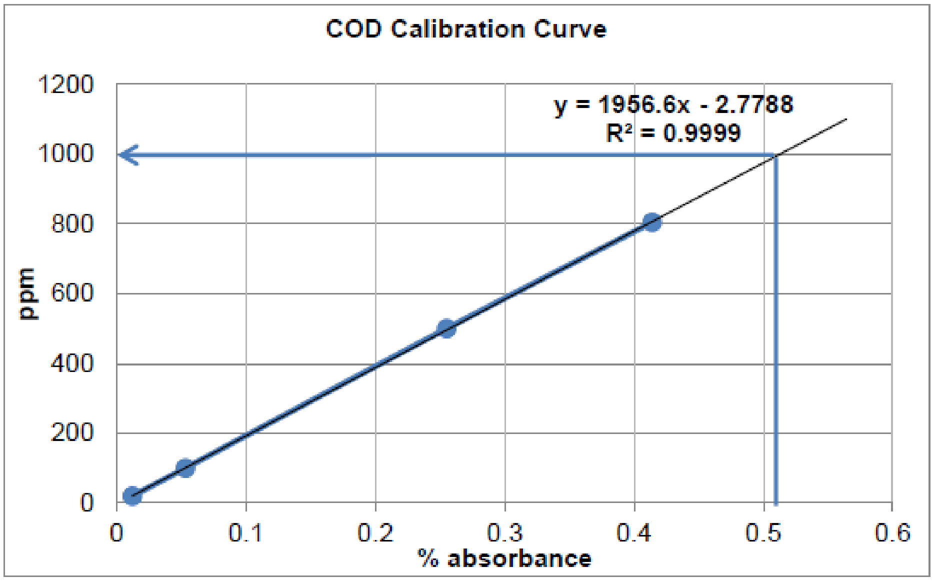


Figure 3-11 COD calibration curve

3.4.6 Reactor Dismantling

After 241 days (almost 8 months) of operations, when the gas production ceased, ten MSW reactors (M1 to M10) were disassembled to assess the physical properties, such as settlement, weight loss, moisture content and volatile solids, of the degraded waste. Reactor F8, the food waste reactor with horse manure, was dismantled after just 70 days of operation because it had not produced any gas. The remaining seven food waste reactors (F1 to F7) were dismantled after 579 days (almost 19 months) of operation, and tests, similar to those performed for the MSW reactors, were conducted. Figure 3-12 shows the dismantling of the reactors. Figure 3-13 shows the degraded waste after dismantling.

(a) (b) (c)

Figure 3-12 (a) Dismantling of reactors; (b) Height measurement; (c) Weight measurement

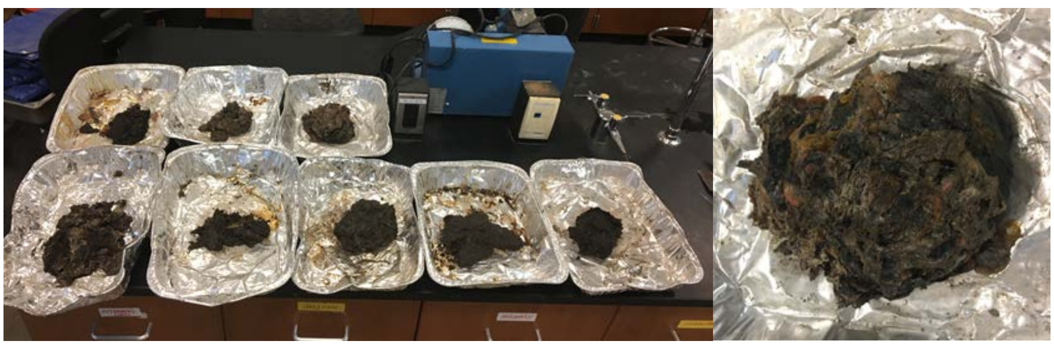


Figure 3-13 Degraded waste after dismantling

3.5 Design Considerations and Construction of Biocell for Field Application

3.5.1 Design Considerations

The planning and design of an effective and efficient biocell require an extensive literature review before its construction, instrumentation, and monitoring. Several detailed designs of biocell were prepared, outlining specific drawing details of every component, feasible dimensions of the field scale cell, construction procedures and steps, intricate details of the instrumentation, and technical justification for critical components. Based on the laboratory scale study, two field test cells (control cell and biocell) were installed in the field. The layout of the field scale test cells is shown in Figure 3-14. The cells were identical and contained the same feedstock, but nutrients were only applied in the biocell only. The components of the cells included a leachate collection and removal system, a gas collection system, and a system to continually monitor the temperature and moisture of the MSW inside the cells. The feedstock, construction, and instrumentation systems are discussed in separate sections. The key features of the field biocell are illustrated in the following section.

- The biocell was a custom-designed waste container made of steel, with dimensions of 21 ft. by 8 ft. by 8 ft. It had a three-well head (Figure 3-15).
- Each cell was equipped with three vertical perforated pipes for gas collection, a horizontal leachate recirculation pipe attached at two sides of the cell, and one leachate collection pipe.
- Three verticals gas wells were connected with a common 4” header pipe for gas collection. To measure the amount of methane production, a gas flow meter (ST100 Mass Flow Meter by FCI), as shown in Figure 3-16 (a), was connected with the header pipe. The flow meter required a solar panel to supply continuous 24 Volt DC (direct currents).
- A layer of gravel covered the bottom of the cell, with a geotextile layer placed over it to provide for drainage of the leachate. The box was tilted on the slope to provide adequate gradient for the leachate to flow to the leachate collection sump pipe.
- Each cell was provided with a leachate reservoir tank. A pressure-activated pneumatic pump was used to extract leachate from the cell and send it to the reservoir. A VP4 bottom-loading pneumatic pump, manufactured by Viridian, was installed in the leachate collection sump, as depicted in Figure 3-16 (b). The pump required an air compressor to supply air to the pump. The leachate collection pipe was provided with a screener and a geotextile separation layer to avoid possible clogs in the pipe.

- Each cell had a lid with three 6-inch ports for gas collection, one 2-inch port for leachate recirculation, and another 2-inch port for removing the sensor cables from the cells.
- Four temperature sensors were installed in each cell and were supported by two layers of PVC pipe frame, which was attached at the side of the box. The sensor port was outside the PVC pipe so that it was in direct contact with the waste, while the sensor wire was inside the pipe, and was connected with a data logger station.

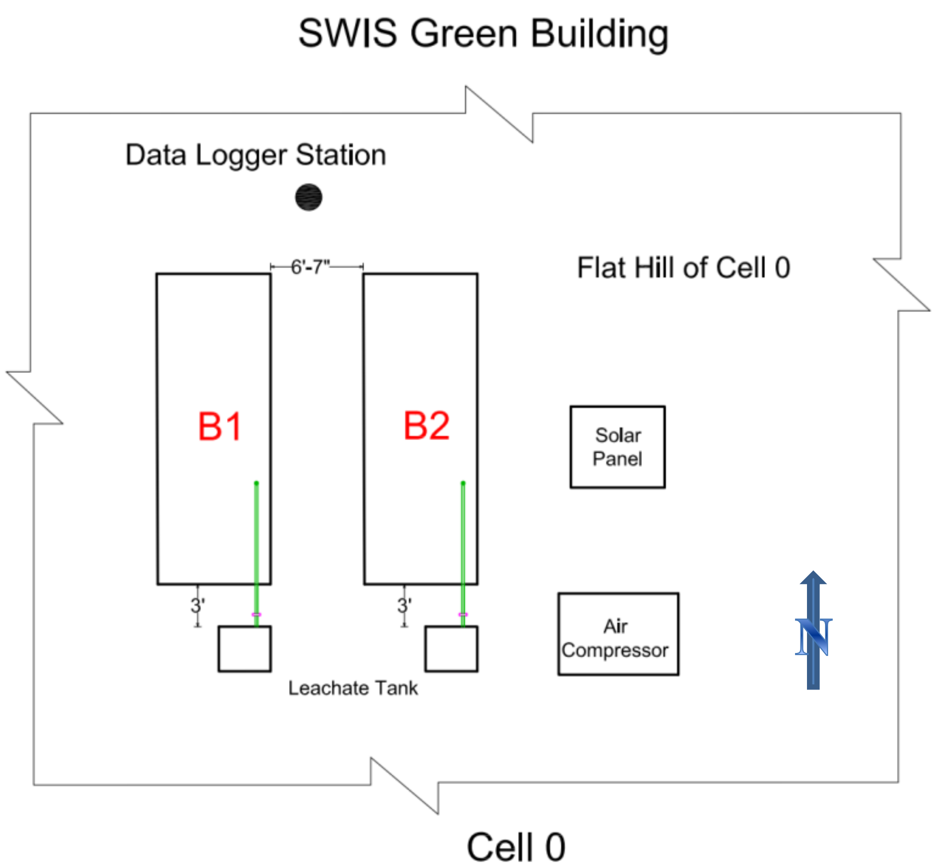


Figure 3-14 Layout of field test sections

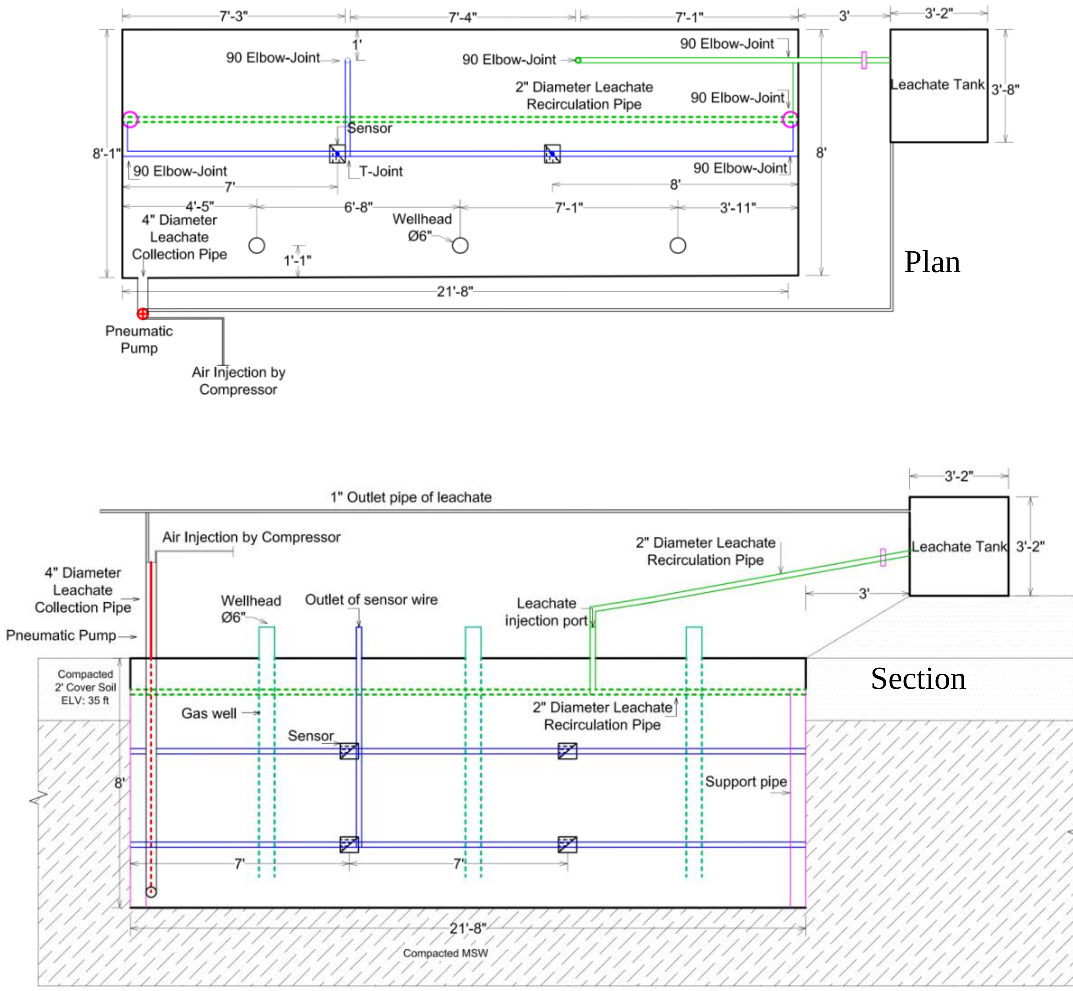


Figure 3-15 Plan and section of biocell

(a) (b)

Figure 3-16 (a) ST100 Mass Flow Meter (FCI); (b) VP4 Bottom-Loading Pneumatic Pump

3.5.2 Combination of Feedstock and Inoculum

Residential solid waste was chosen as the feedstock for the biocell. Residential waste typically contains high organic and moisture content, which improves gas production. Based on the concept of the biocell, only organic waste that had been sorted at the material recovery building at the City of Denton Landfill was deposited in the biocell. Approximately five tons of wastes were deposited in the control cell. Four tons of waste, 0.5 tons of sludge (Class B type biosolids from the waste water treatment plant), and 0.3 tons of pig manure were deposited in the biocell. Waste was not compacted by any equipment; rather was allowed to be compacted by its own weight.

Table 3-2 Combination of feedstock and inoculum for field scale

	Feedstock (MSW)	**Sludge**	**Pig Manure**
B1 (Control Cell)	100%	-	-
B2 (Biocell)	84%	10%	6%

3.5.3 Construction and Instrumentation

The details of biocell construction and installation in the field are discussed in Chapter 5, which describes a number of extensive tasks that were performed to simulate the actual landfill condition in the field.

3.5.4 Monitoring

A significant amount of on-site real-time data was required to address the objectives of this research,. An extensive monitoring program was designed to allow collection of data related to gas production, waste degradation, and biochemical reaction kinetics, as well as to determine the general waste biodegradation characteristics. Analysis of the collected data and specific environmental parameters described in Table 3-3 was performed to fulfil the research objectives.

Table 3-3 Monitoring of environmental parameters

Environmental Parameters	**Monitoring Techniques**
Waste Temperature and Moisture	Sensor
Leachate Quality	pH, COD, BOD
Leachate Quantity	Volume
Gas Composition	Gas Analyzer
Gas Volume	Flow Meter
Waste Characteristics	Physical composition, Moisture content and Volatile solid (VS)
Waste Stabilization	BOD, COD, VS, Settlement

Chapter 4

Experimental Study of Laboratory Scale Biocell

4.1 Introduction

This chapter describes the results and analysis of the experimental program of the laboratory-scale biocell. The chapter is divided into three sections: physical and hydraulic characteristics of the feedstock and inoculum at the beginning of the study, monitoring of reactors during the study, and physical and hydraulic characteristics of the waste at the end of the study. At the beginning of the study, fresh municipal solid wastes were collected from the working phase of the City of Denton Landfill. Food waste was collected from two sources: fruit and vegetable wastes were collected from Walmart, Denton, Texas, and food scrapings were collected from the University Center Cafeteria (Connection Café) at the University of Texas at Arlington. Various tests were performed to measure the physical composition, moisture content, and volatile solids of the waste. Characteristics of leachate and gas of the MSW and food waste reactors were monitored for 8 months and 20 months, respectively. Various tests were performed to measure the pH, BOD, and COD of the leachate, as described in Chapter Three. Volumes of the gases generated from the reactors were measured, and their composition was characterized as well. MSW reactors and food waste reactors were dismantled after 241 days and 579 days of operation, respectively, and various tests were performed. The final moisture content, volatile organic content, and weight and height loss of feedstock were compared with the initial condition to determine the degree of decomposition or waste stabilization in the reactors.

4.2 Properties of MSW and Food Waste

4.2.1 Physical Composition

Both the MSW and food waste samples were sorted manually (hand sorting), and their physical composition was determined by wet weight basis. The MSW sample was composed of 34% paper, 19% plastic, 13% food waste, 8% textile, 2% styrofoam and sponge, 9% yard and wood waste, 3% metals, 2% glass, 4% construction debris, and 6% others (soils and fines). The food waste from the University cafeteria consisted of 49.85% fruits and vegetables, 30.15% grain products, 9.98% meat and seafood, and 2.02% dairy products. Results obtained from the physical composition test of MSW samples and food waste samples are presented in Figure 4-1 and Figure 4-2, respectively.

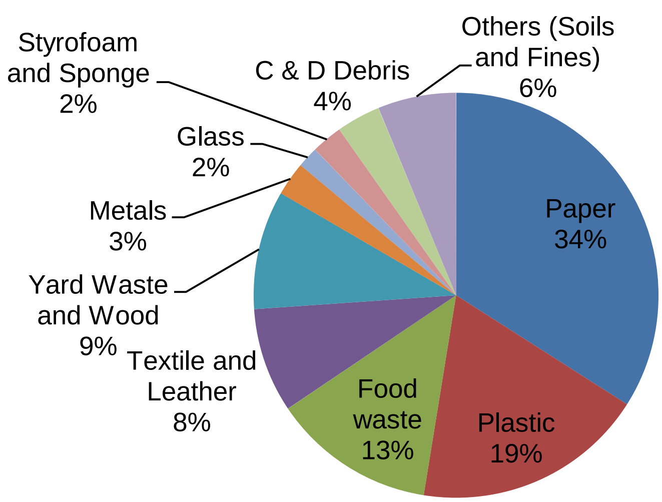


Figure 4-1 Average physical composition of MSW

Gas production from waste depends on the degradable portion of the entire waste mass. As food waste is completely organic, it is 100% degradable. MSW samples consist of both degradable and non-degradable portions; therefore, the fraction of the degradable component of MSW was determined on a wet weight basis. MSW samples were classified based on their degradability, as presented in Figure 4-3.

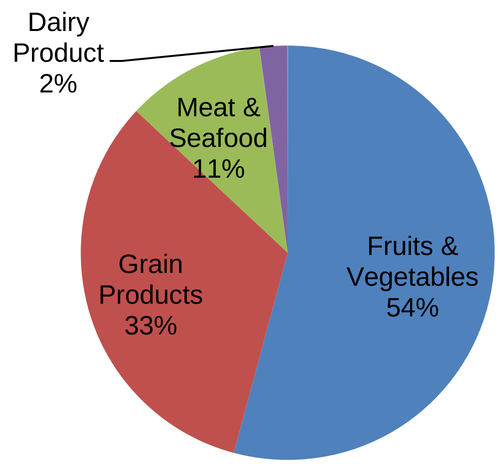


Figure 4-2 Average physical composition of food waste

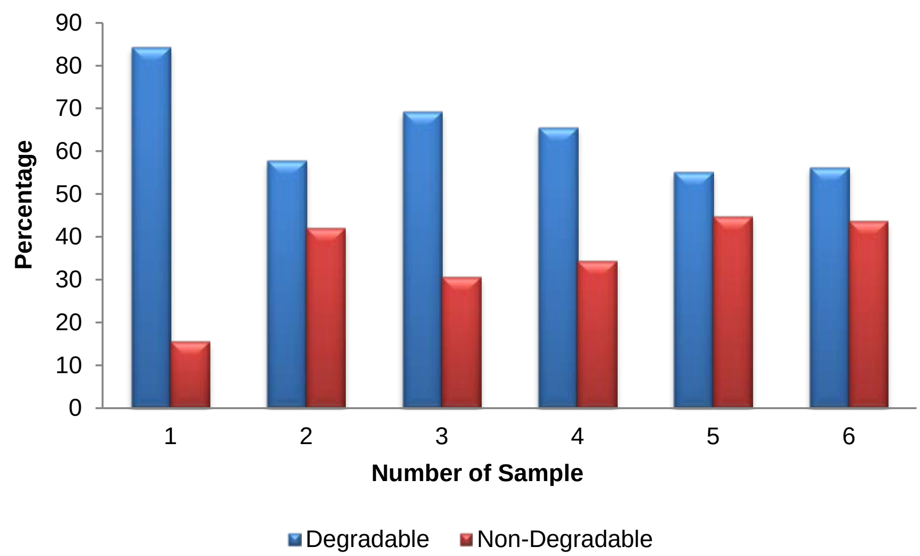


Figure 4-3 Physical composition based on degradability of MSW

It was found that about 64.74% of the MSW was degradable, while the remaining 35.26% was non-degradable. The degradable portion was composed of food waste, paper, yard and wood waste, and textile; while non-degradable waste was plastic, styrofoam and sponge, metal, glass, construction debris, and other. Figure 4-4 shows that the MSW samples consisted of 51.47% paper, 21.88 % food waste, 12.83% textiles and leather, and 13.8% yard and wood waste.

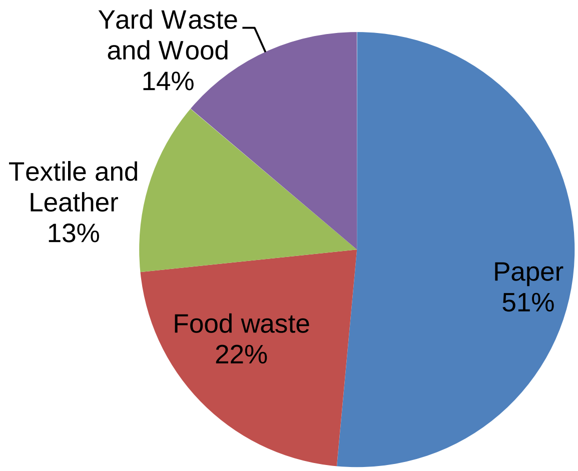


Figure 4-4 Physical composition of degradable components of MSW

Physical composition of MSW determined in the current study was compared with studies conducted by the US EPA (2014) and by Taufiq, T (2010), and the results are presented in Table 4-1. The physical composition of MSW varies considerably, based on the regional, cultural, social, and economical influences (Denafas et. al., 2014). It is also time dependent; therefore, it is likely that the waste composition will vary at the same landfill at different times of the year. Country averages also vary from city to city. In this study, the percentage of paper was lower than in the study conducted by Taufiq, T (2010), although both studies were conducted on the City of Denton Landfill. The percentage of food waste percentage in this study is significantly higher than that reported by Taufiq, T (2010), but close to the US national average, per the US EPA (2014). The percentage of plastic in this study is almost identical to that reported by Taufiq, T (2010), but more than the national average. Overall, the average composition of degradable and non-degradable waste in this study was similar to the national average in 2014.

Table 4-1 Comparison of physical composition of fresh MSW

Components	US national average (US EPA, 2016) %	Taufiq, T (2010) %	Current study, %
Paper	26.6	40	33.97
Plastic	12.9	18	18.47
Food waste	14.9	2	12.98
Textile	9.5	4	8.33
Yard trimming	13.3	9	9.46
Wood waste	6.2		
Metal	9	5	2.69
Glass	4.4	1	1.67
Styrofoam	-	1	2.46
C&D debris	-	2	3.51
Others	3.2	18	6.25
Total	100	100	100
%Degradable	70.5	55	64.74
%Non-degradable	29.5	45	35.26

According to ReFED (2016), the national food waste composition is 42% fruits and vegetables, 19% grain products, 14% meat and seafood, and 26% dairy products (Table 4-2). In the current study, other than dairy products, the components were similar to the national average. Table 4-3 depicts the composition of MSW and food waste in the reactors. According to the physical composition of degradable waste shown in Figure 4-4, only organic wastes were used in MSW reactors. The breakdown of the total MSW shows that paper represents the largest component at 50%, followed by food waste of 20%, yard waste of 15%, and textiles of 15%. The national waste composition of food waste of USA (United States Department of Agriculture) was used in food waste reactors where fruit and vegetable is 50%, meat and fish is 20%, grain products is 20% and dairy products is 10%.

Table 4-2 Comparison of physical composition of food waste

Components	US national average (ReFED (2016), %	Current study, %
Fruits and Vegetables	42	49.85
Grain Products	19	30.15
Meat and Seafood	14	9.98
Dairy Products	26	2.02

Table 4-3 Composition of MSW and food waste in the reactors

MSW		**Food Waste**	
Type	**Composition**	**Type**	**Composition**
Paper	50%	Fruit and Vegetable	50%
Food Waste	20%	Meat and Fish	20%
Yard Waste	15%	Grain Products	20%
Textile	15%	Dairy Products	10%

4.2.2 Moisture Content

Moisture content tests were performed on the MSW samples according to the procedure described in Section 3.4.5.1 in Chapter Three. The water content is the ratio of "pore" or "free" water in a given mass of soil to the dry or wet solid waste. It is expressed as a percentage. The moisture content was determined on both dry and wet weight basis; however, the moisture content for MSW is expressed as wet weight basis only. During the physical composition tests, the moisture content of the fresh waste samples was determined in both wet weight and dry weight basis. The initial moisture content (wet weight basis) for each reactor is listed in Table 4-4.

Table 4-4 Initial moisture content of MSW and food waste in the reactors

Reactors	**Wet wt. (lb.) in the reactor**	**Moisture content (%) (wet weight Basis)**	**Dry wt. (lb.)**	**Total Solid, TS (%)**
M1, M2	2	26.61	1.47	73.39
M3, M4	2	26.58	1.47	73.42
M5, M6	2	25.70	1.49	74.3
M7, M8	2	27.32	1.45	72.68
M9, M10	2	27.43	1.45	72.57
F1, F2	4	76.67	0.933	23.33
F3, F4	4	75.73	0.971	24.27
F5, F6	4	77.66	0.894	22.34
F7, F8	4	76.15	0.954	23.85

The average moisture content of fresh MSW was 26.7% on wet weight basis in the MSW reactors. The average moisture content of fresh food waste was about 76.55%

on wet weight basis. The moisture content in the reactors was influenced by the moisture content of the waste feedstock, the moisture content of the inoculum, and the addition of water during the filling and compaction of the feedstock in the reactors. Though similar types feedstock were used in the reactors, due to the presence of different kind of manures (cow, horse, and pig), moisture content in the reactors varied to some extent. Because the MSW reactors contained less moisture (26.55%) than the food waste reactors (76.55%), extra water was added to each of the MSW reactors during waste filling. The additional water also ensured proper compaction of the waste and likely created an ideal ambient environment for proper microbial activity. The MSW reactors had total solids of 73.27%; food waste reactors had only 23.45%, which affected the amount of gas generation in the reactors. Excessive moisture content may hinder the gas production.

4.2.3 Volatile Solid Content

According to the Interstate Technology and Regulatory Council (2006), the volatile solid test is the most inexpensive measurement of the amount of biodegradable material that remains in the waste mass. Volatile organic content indicates the amount of organic materials in the waste mass. The Volatile organic content tests were conducted twice on the feedstock samples: before the reactors were sealed (initial volatile solid content) and after the end of the study, as described in Chapter Three. The initial volatile solids of the MSW and food waste reactors are listed in Table 4-5.

Volatile solids accounted for about 85.9% of the MSW feedstock and 92.16% of the food waste feedstock. According to Taufiq (2010), the volatile organic content of fresh MSW is usually about 76.96%. The higher percentage of volatile solids in this study was due to the use of organic MSW (paper, food waste, textiles, and yard waste) in the reactors. Food waste has a higher gas generation potential because of the higher volatile solid content. Due to the high moisture content in the food waste reactors in this study, the total solid content was less in the food waste reactors than in the MSW reactors. As a result, the amount of volatile solids was less in the food waste reactors than in the MSW reactors, even though food waste has a higher volatile organic content. The MSW reactors contained 1.328 lbs. of volatile solids, which is almost 2.8 times more than the amount in the food waste reactors (0.47 lbs.). Volatile solid contents are important because they affect the amount of gas production.

Table 4-5 Initial volatile organic content of MSW and food waste

Reactors	**Inoculum**	**Volatile Solid, VS (%)**	**VS (lb.)**
M1, M2	-	86.71	1.35
M3, M4	Cow Manure, 6%	84.41	1.29
M5, M6	Pig Manure, 6%	87.04	1.35
M7, M8	Horse Manure, 6%	86.40	1.34
M9, M10	MnP Enzyme, 0.00000213%	84.96	1.31
F1, F2	-	91.66	0.50
F3, F4	Cow Manure, 6%	92.96	0.48
F5, F6	Pig Manure, 6%	91.78	0.46
F7, F8	Horse Manure, 6%	92.23	0.44

4.3 Properties of Inoculum

Inoculum is a source of microorganisms and plays a vital role in waste degradation and methane production. Literature cites examples of inoculum being used to enhance waste degradation, such as in sewage treatment sludge, animal manure, cellulose and lignocellulose enzymes, and old landfill leachate (Karanjekar, 2013; Al-Kaabi et al., 2009; Callaghan et. al., 2002; Lopes et. al., 2003; Sosnowski et. al, 2003; Sah, 2006; Cirne et al., 2008, Jayasinghe et.al., 2013; an Erses and Onay, 2003). The addition of inoculum shortens the duration of the waste degradation process and increases gas production significantly. Though previous researchers used higher amount of inoculum in their laboratory experiments, smaller amounts of manure (6%), Manganese Peroxidase (0.00000213%), and sludge (10%) were used in this study to check the feasibility for field application. Two major properties of inoculum such as moisture content and pH were measured in this study. The sludge used had a high moisture content (72% in wet weight basis) and high pH (8.37), which meant that it was rich in anaerobic microorganisms. The moisture content age of the three different manures varied significantly, as they were obtained from different sources. The cow and pig manures had very low moisture content (2% and 5 %, respectively) as they were 9-12 months old, while the horse manure had moisture content of 36% and was less than a week old. The pH of the cow, pig, and horse manure were about 8.95, 7.81, and 7.69, respectively, as shown in Figure 4-5. The high pH of the inoculum used in this study helped to neutralize the acidic environment, and hence reduced the acidogenic and transition phases and started the methanogenic phase early.

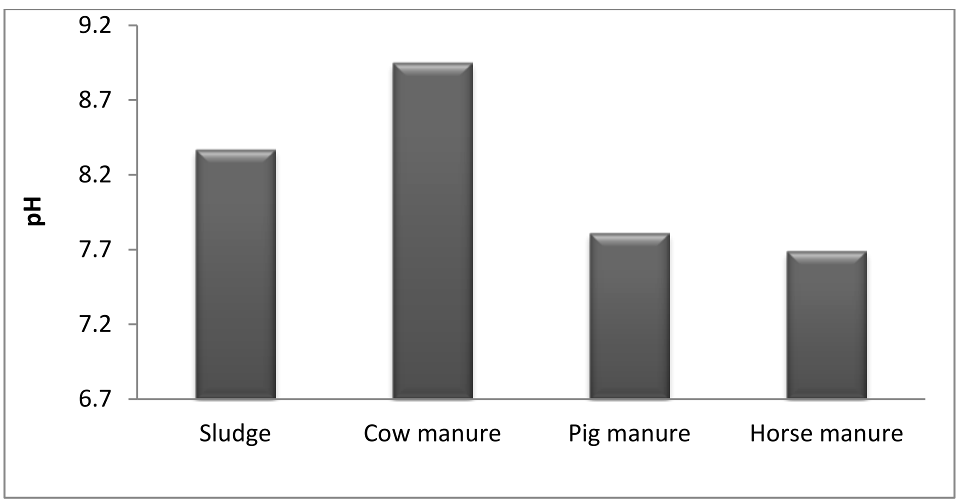


Figure 4-5 pH of sludge and manures

4.4 Gas Characteristics

Landfill gas, or biogas generation, is the main indicator of waste degradation. In this study, composition, volume, and rate of gas generated from the MSW reactors (M1 to M10) and food waste reactors (F1 and F8) were measured on a regular basis, as described in the section below:

4.4.1 Gas Composition of MSW Reactors

Landfill gas mainly consists of methane, carbon dioxide, and oxygen, with traces of other gases such as nitrogen, hydrogen sulfide, ammonia, and non-methane organic compounds. The Landtec GEM 2000 was used in this study to measure the percentages of all of the different components of the gas generated in the reactors. After the reactors were installed, the first gas was measured on day 9 in MSW reactors M1 to M10. At the beginning, the carbon dioxide content was high and the methane content was low in all of the MSW reactors. The oxygen content was very low from the beginning due to the small size of the reactors; the amount of trapped oxygen during installation was also very low. As a result, the aerobic phase of the waste degradation process ended early, and the acidogenic phase commenced early. In the acidogenic phase, degradable organic compounds break into simpler compounds, such as carbon dioxide and water vapor. The carbon dioxide content peaked on day 9 and reduced with time, as the methane content rose. Other gases also decreased with time; however the oxygen content remained steady at 2%-3%. The methane content was below 2% for 20 – 25 days, and the pH of the leachate dropped below 5.5 in all of the MSW reactors due to the accumulation of volatile fatty acids (VFA). On the 25th day of operation, 10% of sludge was added to all of the MSW reactors during leachate recirculation, which helped to neutralize the acidic environment inside the reactors and start the methane production in reactors M5 and M6 with pig manure and reactors M9 and M10 with manganese peroxidase. Other MSW reactors started producing methane after 33 to 41 days. During the transition phase, between the acidogenic and methanogenic phases, the percentage of carbon dioxide reached 40% of the total gas composition. As soon as the methanogenic phase started, the methane-to-carbon-dioxide ratio (CH_4:CO_2) began expanding, and the volume of other gases decreased. During the methanogenic phase, the methane content in all

MSW reactors reached 60%-65%, which was also seen in a study by Karanjekar (2013). Figure 4-6 shows the variations of methane percentages with time for MSW reactors M1 to M10. The highest methane contents seen were 66.20% and 66.10% in the reactors with pig manure (M5 and M6, respectively), followed by the reactors with cow manure (M3 and M4). Reactors with MnP had methane contents of 64.1% in M9 and 62.6% in M10. The lowest methane content was seen in the control reactors, M1 and M2, which was 52.3 and 60.1%, respectively.

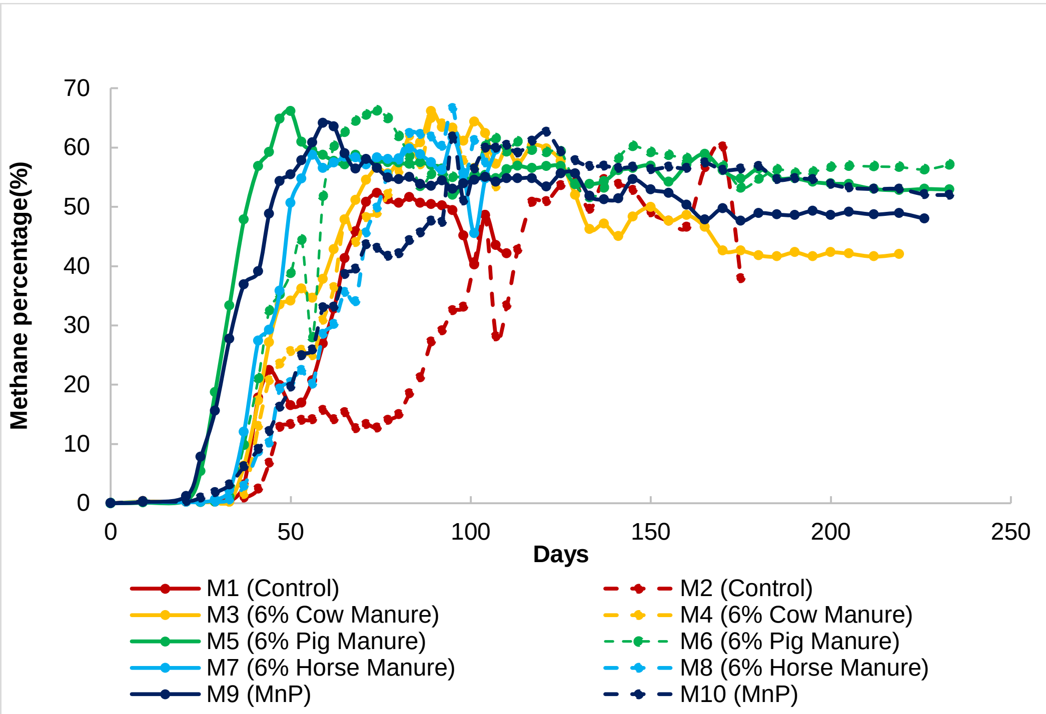


Figure 4-6 Methane content in MSW reactors

Figure 4-7 shows the ratio of methane to carbon-dioxide (CH_4:CO_2) in the MSW reactors. The carbon dioxide content was as high as 60% from the beginning, and the methane content was very low. Hence, the ratio of CH_4 to CO_2 was almost zero. With time, the percentage of carbon dioxide began to diminish, while the percentage of methane began to increase after the acidogenic and transition phases were completed and the methanogenic phase begun. As a result, the ratios of CH_4 and CO_2 also increased with the increase of methane content. During the methanogenic phase, the ratios of CH_4 and CO_2 for all of the reactors ranged from 1.3 to 2.8.

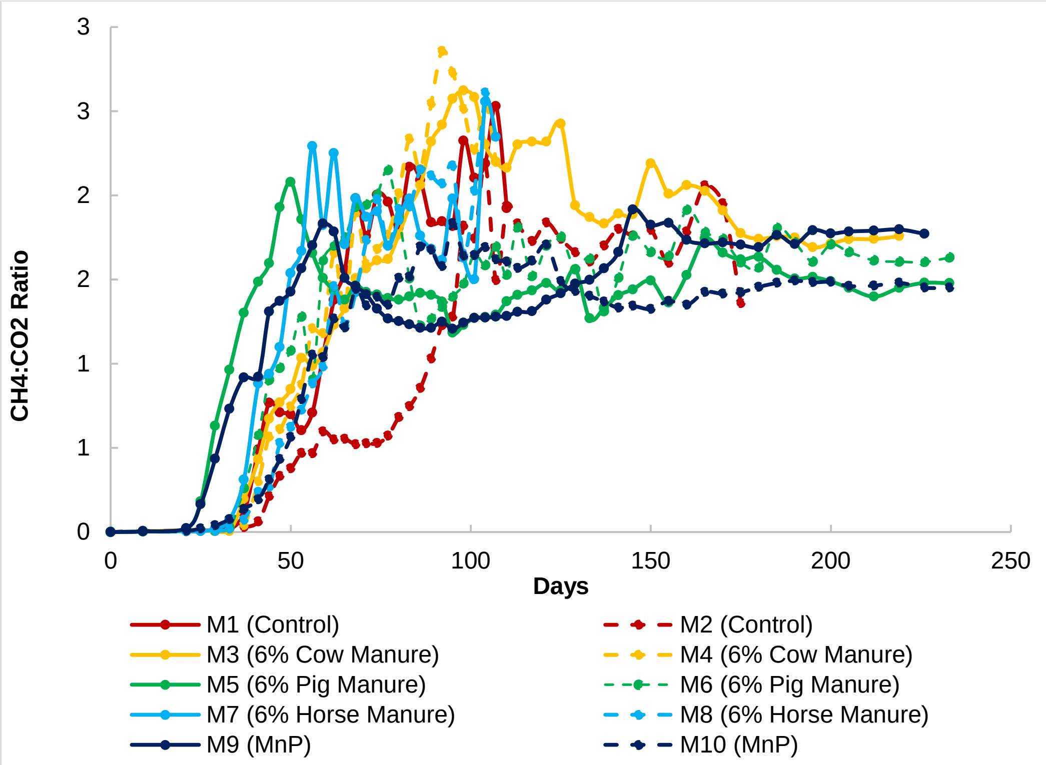


Figure 4-7 Methane-to-carbon-dioxide ratio in MSW reactors

Figure 4-8 shows the percentage of anaerobic activity in the MSW reactors, which is dependent on the concentrations of methane and carbon dioxide in the landfill gas. The concentrations of the CH_4 and CO_2 gases can be used to estimate the fraction of waste that degraded anaerobically at any point in time. Landfill gas is usually composed of 45-60% CH4 and 40-60% CO2 (Tchobanoglous et al., 1983). Based on the stoichiometry of the reactions of aerobic and anaerobic degradation, the percentage of waste degraded anaerobically, P, can be estimated by the following equation developed by Yazdani (2010),

$$P = \frac{2C_{CH_4}}{2C_{CH_4} + (C_{CO_2} - C_{CH_4})} \times 100 \text{ .... (Eq. 4.1)}$$

Where C_{CH4} and C_{CO2} are the measured concentrations (% v/v) of CH4 and CO2, respectively.

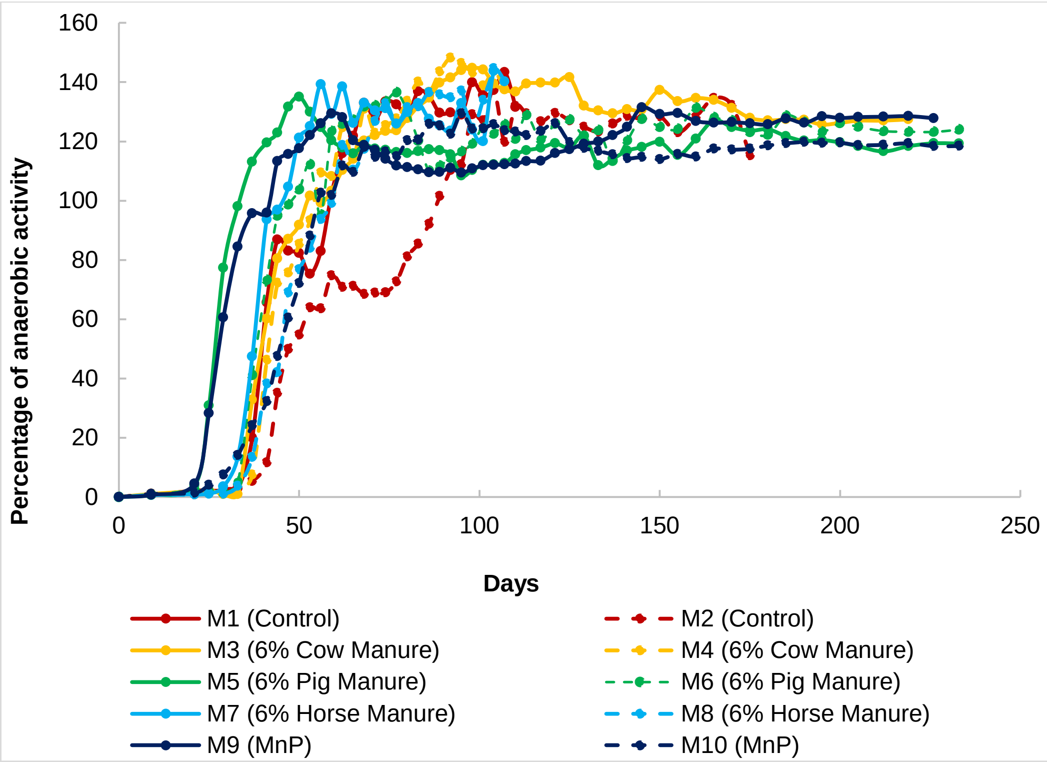


Figure 4-8 Percentage of anaerobic activity in MSW reactors

The percentage of anaerobic activity in MSW reactors varies from 0% to 148%. Anaerobic activities increase with an increase of methane content and a reduction of carbon dioxide content. Except for one control reactor, all of the MSW reactors achieved more than 100% of anaerobic activity after 60 days.

4.4.2 Gas Composition of Food Waste Reactors

The first measurement of gas volume and composition was made on day 4 from food waste reactors F1 to F8. In the beginning, the oxygen and methane content were low, and the carbon dioxide content was very high. With the beginning of the acidogenic phase, the pH began dropping due to an excessive accumulation of volatile fatty acids. Although the leachate generated from the reactors was neutralized by potassium hydroxide (KOH) and was recirculated frequently, the food waste reactors went into a long lag phase. After 45 days of operation, the food waste reactors started producing very small amounts of methane again. In contrast, methane production was considerable in the MSW reactors, after just 25 days. Since, very low amount of methane production was observed in the food waste reactors for a considerable period of time, reactor F8 was dismantled on day 70 to investigate the reason for the lag phase which can be explained by the pH of leachate of food waste reactors. Again, all of the food waste reactors except F3 stopped producing methane after 91 days. Reactor F3, with cow manure, reached 40% methane production after 110 days of operation, whereas the other food waste reactor, F4, exceeded 40% on day 144. One of the horse manure reactors, B7, reached 40% methane production after 260 days; the other reactors, F1; F2; F5; and F6, took more than 300 days to produce 40% methane. Figure 4-9 shows the methane content in the food reactors. The food waste reactors with cow manure (F3 and F4) produced more than 70% methane for 150 days. The maximum methane content was 76.10% in reactor F3 on day 198. All of the food waste reactors except F8 achieved more than 70% methane during operation, which is significantly higher than the methane content in a landfill and is similar to the methane content in an anaerobic digester. All of the reactors, except the control reactor (F2), stopped producing gas after 494 days.

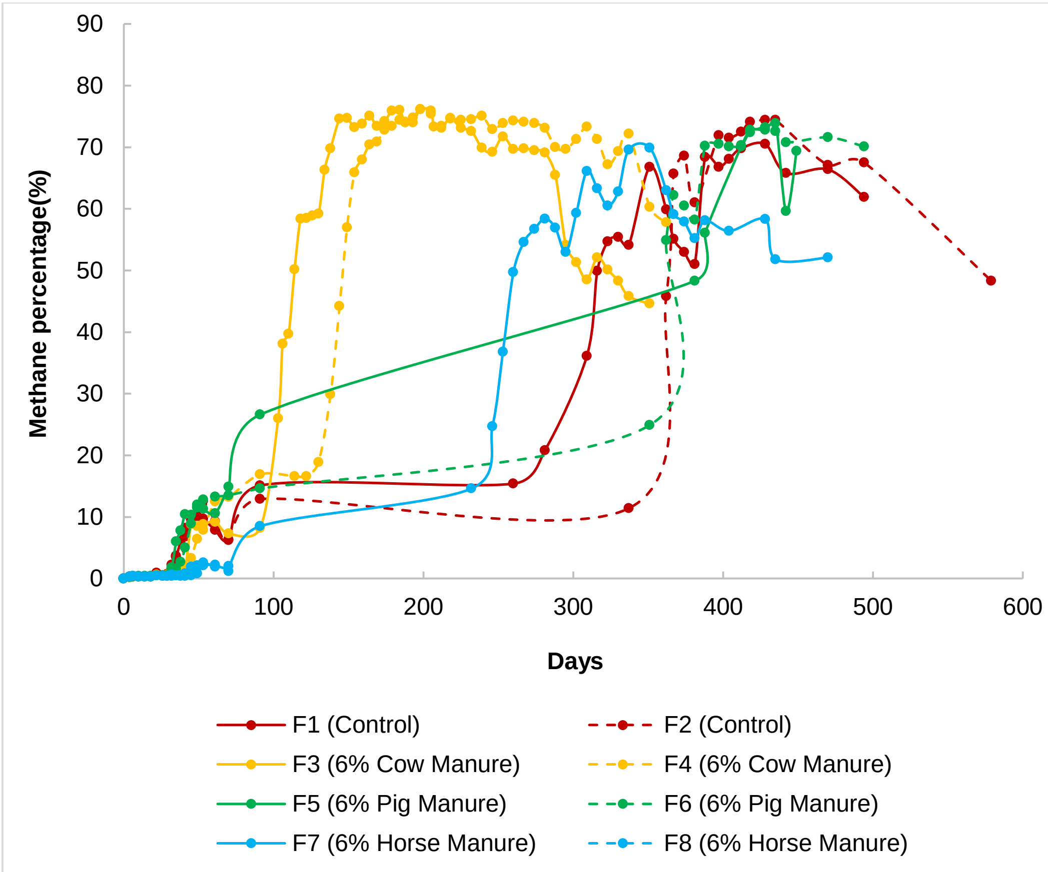


Figure 4-9 Methane content in food waste reactors

In the methanogenic phase, the percentage of methane increases as high as 60% to 65%, with the variation of pH of the leachate from 6.0 to 8.5 (Karanjekar, 2013). Carbon dioxide content reduces, which can be seen from the increase of methane to carbon dioxide ratio (CH_4:CO_2 ratio). In this study, after 103 days of operation, the CH_4:CO_2 ratio in reactor F3 reached 1.5%. Figure 4-10 shows the methane-to-carbon-dioxide ratio in food reactors. Reactor F4 with cow manure had the highest CH_4:CO_2 ratio (almost 10) after 362 days.

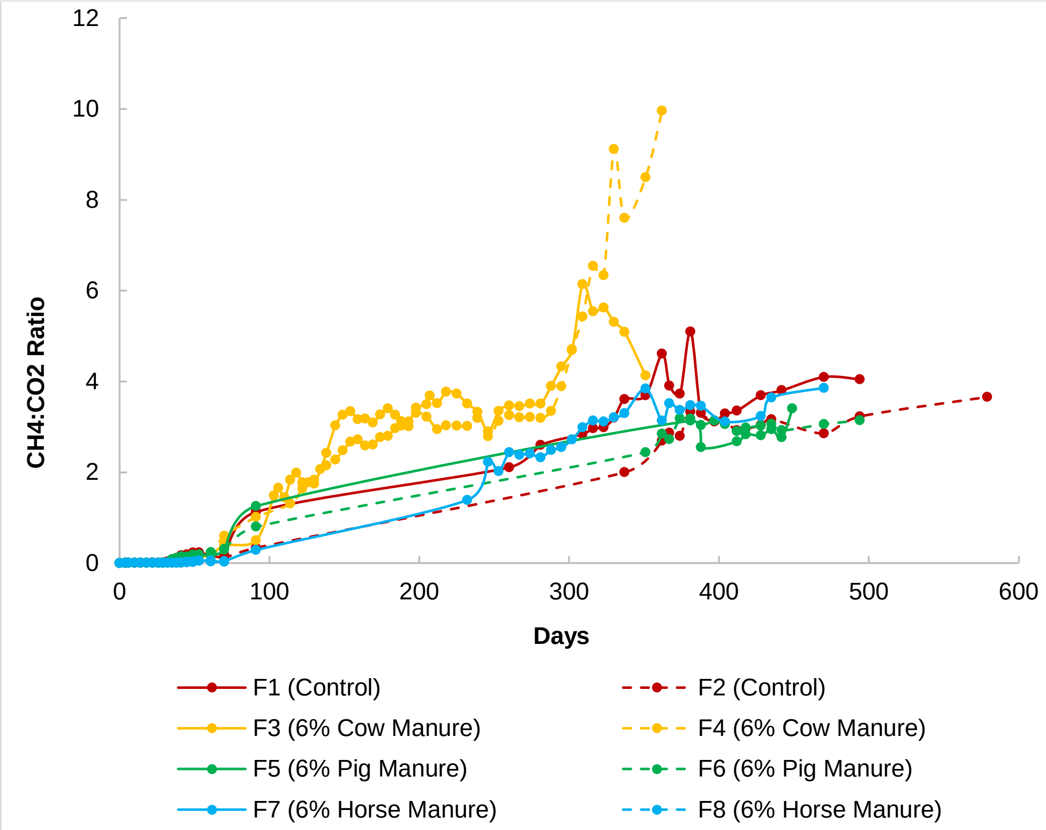


Figure 4-10 Methane-to-carbon-dioxide ratio in food waste reactors

Figure 4-11 shows the percentage of anaerobic activity in food reactors, as calculated by using Eq. 1 described in Section 4.4.1. The value of percentage of anaerobic activity (P) was plotted against time to observe the variations of anaerobic activity in the reactors. After 91 days of operation, only three reactors (F1, F4, and F5) had achieved 100% anaerobic activities; other reactors fell behind due to a long lag phase. Reactors F1 and F5 stopped producing gas after that. In the food waste reactors, the percentage of anaerobic activities varied from 0% to 182%, but overall, the performance of the food waste reactors was inferior to MSW reactors in terms of methane production. Consequently, it can be inferred that the potential for methane production is high for food waste if the lag phase can be reduced.

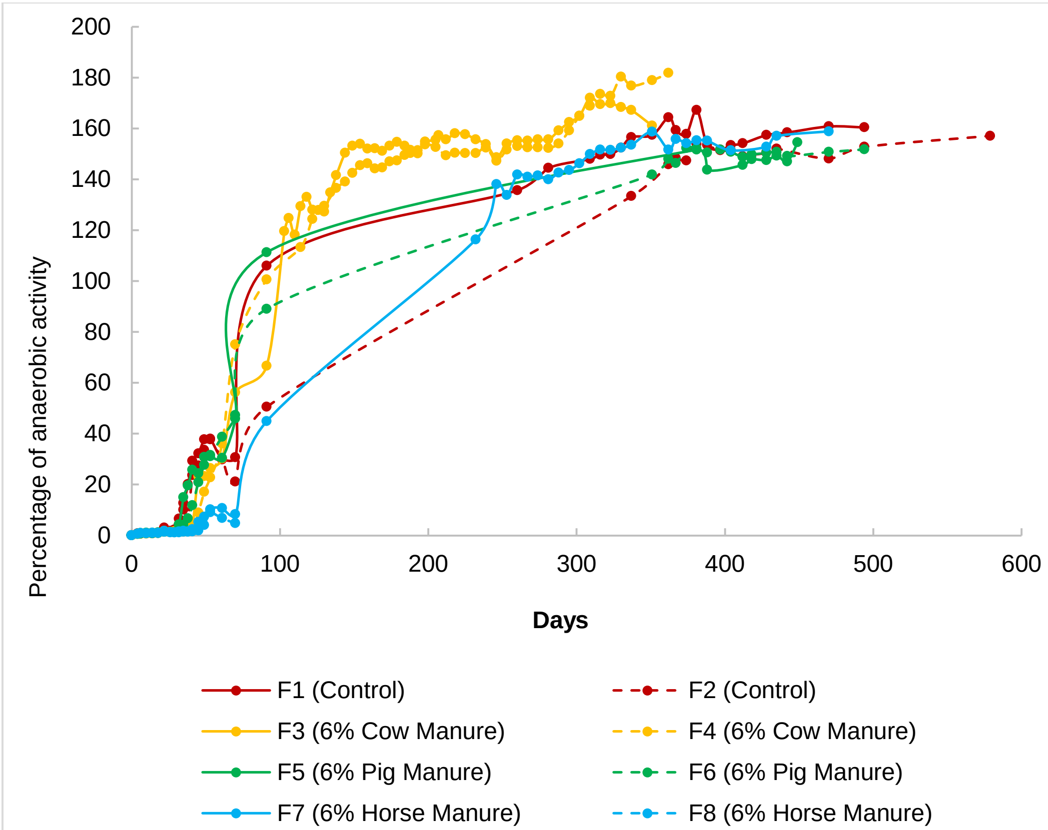


Figure 4-11 Percentage of anaerobic activity in food waste reactors

4.4.3 Gas Volume of MSW Reactors

Landfill gas, or biogas, is the core indicator of waste degradation in a landfill. Biogas production and waste stabilization during anaerobic digestion are affected mostly by moisture content, volatile solids content (VS), and nutrient contents, i.e. amount of inoculum/sludge, particle size, and biodegradability. The biogas or methane yield can be measured by the amount that can be produced per unit of volatile solids contained in the feedstock, after subjecting it to anaerobic digestion for a sufficient amount of time at a given temperature (Zhang et al., 2006).

In this study, MSW reactors produced trivial volumes of gas for the first time on day 8. Sludge was added on day 26 and helped to remove the lag period so that all of the

MSW reactors started to generate a significant amount of gas. Shortly after the sludge was added, the methanogenic phase was attained. This can be clearly observed in the cumulative gas generation graph in Figure 4-12. Reactors with MnP (M9 and M10) produced the highest amount of gas in 233 days (100.6 L/lbs. and 105.1 L/lbs., respectively). The reactors with pig manure (M5 and M6) produced about 85.4 L/lbs. and 83.3 L/lbs., respectively, in 233 days, which was the second highest. The reactors with horse manure (M7 and M8) did not produce a significant amount of gas because of the age of horse manure. The control reactors (M1 and M2) produced a total of only 10.3 L/lbs. and 24.1 L/lbs. gas, respectively. Other than reactors with MnP (M9 and M10), pig manures (M5 and M6), and cow manure (M3), all of the reactors stopped producing gas early and reached steady state phase. Reactor M3 produced 53.6 L/lbs. of gas in 219 days before it stopped generating gas.

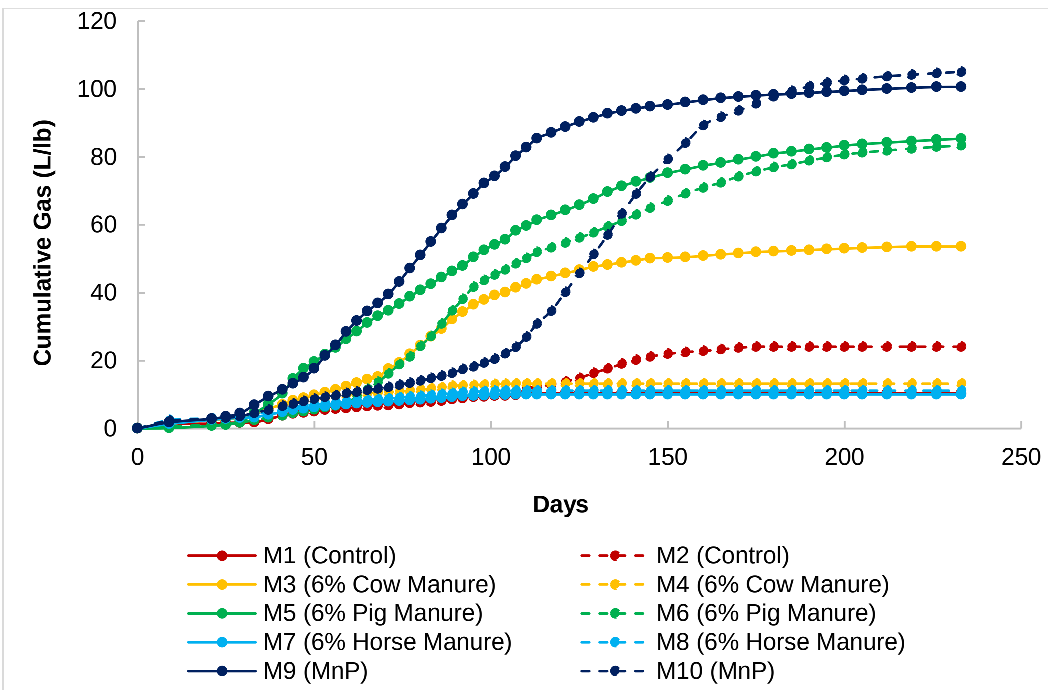


Figure 4-12 Cumulative gas generation (L/lb.) in MSW reactors

Methane production from the MSW reactors was calculated from the percentage of methane content and volume of gas. The cumulative methane versus time graph of MSW reactors follow a trend similar to that of the gas versus time graph shown in Figure 4-13. Reactors M9 and M10 generated about 54.5 L/lbs. and 52.03 L/lbs. of methane, respectively, in 233 days which was the highest among all of the MSW reactors and almost 22 times higher than the control reactor, M1. Reactors with pig manure (M5 and M6) also performed well in terms of methane production from the organic fraction of MSW, which was about 47 L/lbs. and 45.9 L/lbs., respectively, in 233 days. Although the reactors with MnP exhibited the maximum methane yield for the organic fraction of MSW, it is not recommended for field application, as it is expensive. The reactors with pig manure revealed that it can be as productive as the MnP; moreover, pig manure is readily obtainable at an affordable rate. Therefore, it is much more effective and economical to use the pig manure as the additive for field scale biocell operations.

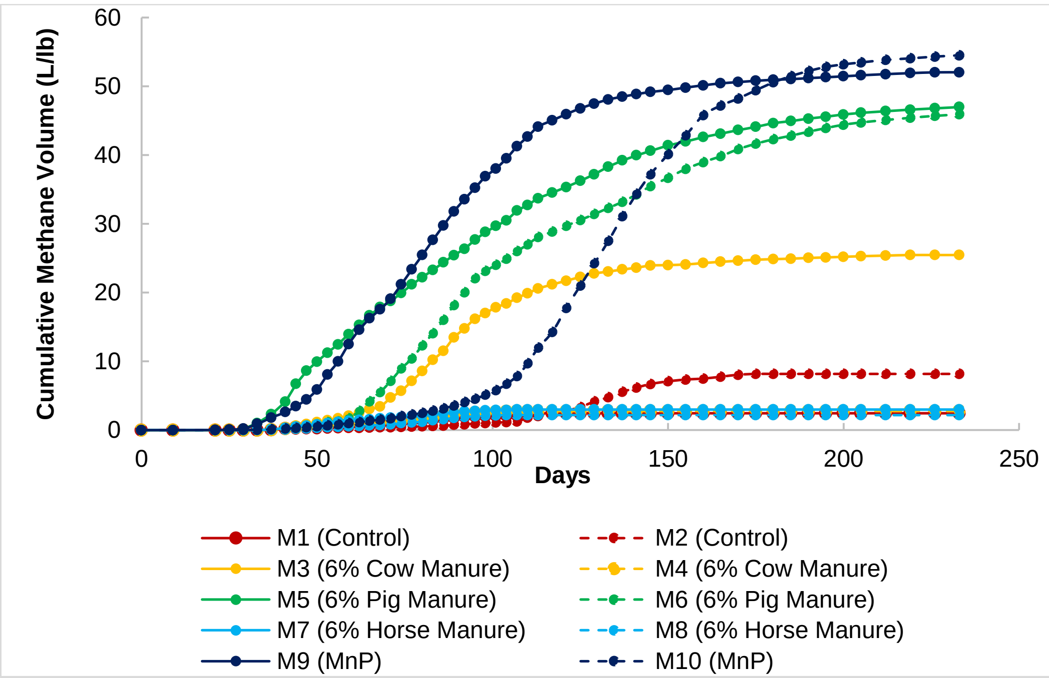


Figure 4-13 Cumulative methane generation (L/lb.) in MSW reactors

The rate of gas generation (gas yield) or methane generation (methane yield) is a cardinal indicator for landfill gas and follows a similar trend in all cases of MSW decomposition. The trend is that the gas or methane yield increases with time before it reaches a peak, then it decreases before ceasing. It may have multiple peaks in its lifetime. From the gas generation versus time plot of MSW reactors, it is clear that all of the reactors experienced similar peak-drop cycles during the monitoring period. The earlier the reactor reaches its peak gas generation rate, the earlier it reaches the methanogenic phase. Figures 4-14 and 4-15 present the rates of gas and methane generation of MSW reactors, respectively.

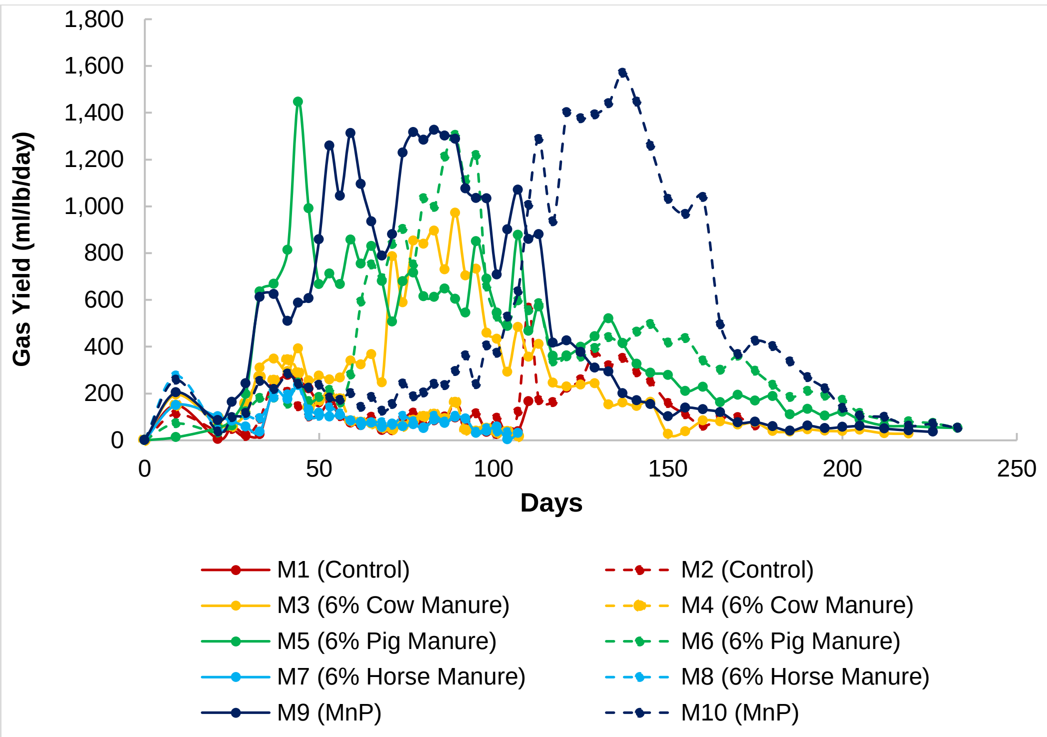


Figure 4-14 Gas yield (mL/lb./day) in MSW reactors

One MSW reactor with pig manure (M5) achieved its peak on day 44, which was the earliest of all the MSW reactors and about 1,447 mL/lbs./day. Another MSW reactor with pig manure (M6) got its peak on day 89, with 1,304 mL/lbs./day. Reactor with MnP (M9) got multiple peaks on days 59 and 83, with 1,312 mL/lb./day and 1,326 mL/lb./day respectively. Another reactor with MnP (M10) achieved its peak on day 137, with 1,569 mL/lb./day which was the highest gas yield among all the reactors. Reactors M9 and M10 had their highest methane yield (841 mL/lb./day and 893 mL/lb./day) on days 59 and day 137, respectively. Reactor M5 had its highest methane yield on day 44 (856 mL/lb./day). MSW control reactors (M1 and M2), one cow manure reactor (M4), and horse manure reactors (M7 and M8) generated very low amounts of gas.

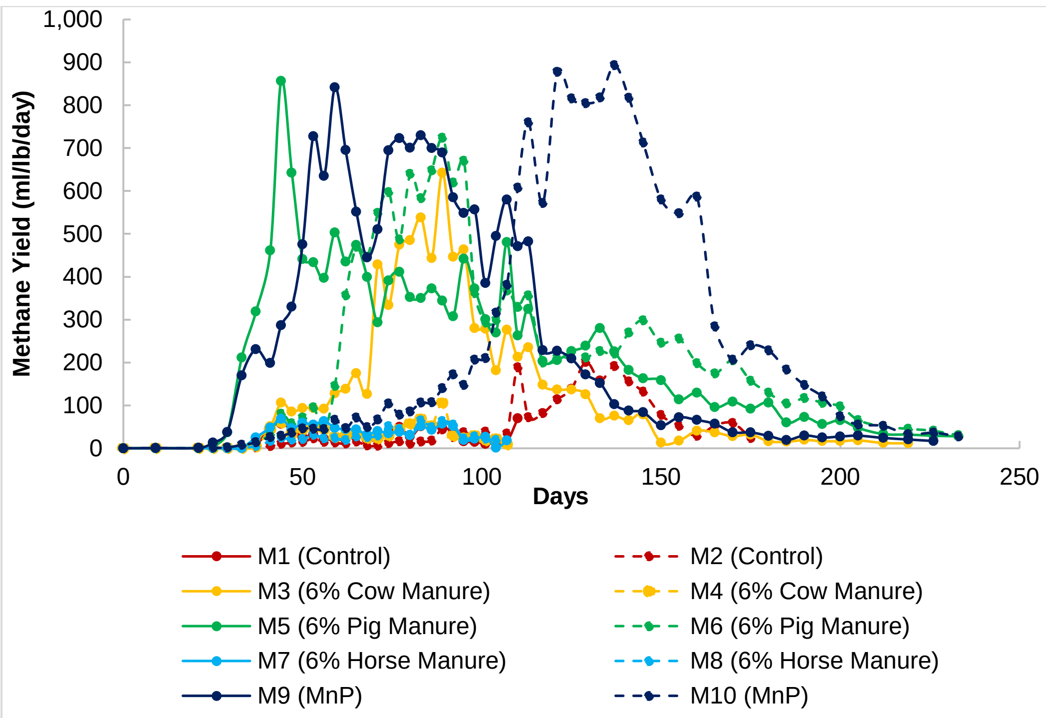


Figure 4-15 Methane yield (mL/lb./day) in MSW reactors

4.4.4 Gas Volume of Food Waste Reactors

In the food waste reactors (F1 to F8), the first gas volume was measured on day 4; it had a very high carbon dioxide content and very low methane and oxygen content. Although the food waste reactors began producing gas earlier than the MSW reactors, over time, they produced smaller volumes of gas and methane than the MSW reactors. Sludge was added to all of the food waste reactors to eliminate the early lag phase. After receiving the sludge, all of the reactors began producing comparable amounts of gas. However, after three months of operation (91 days), all of the food waste reactors except F3 and F4 (with cow manure) entered into a second lag phase and stopped producing gas. Figure 4-16 shows the cumulative gas generation (L/lb.) in the food waste reactors (F1 to F8).

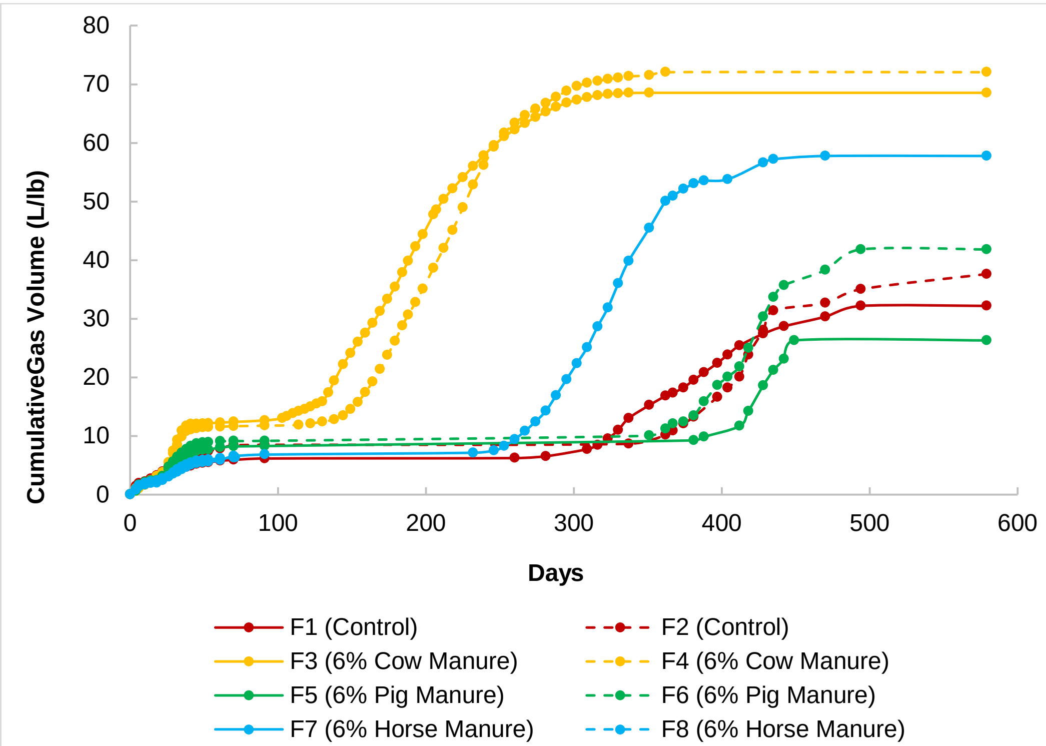


Figure 4-16 Cumulative gas generation (L/lb.) in food waste reactors

The food waste reactors with cow manure (F3 and F4) produced significant amounts of gas (68.5 L/lb. and 72 L/lb., respectively) in about 364 days. MSW waste reactors with cow manure (M3 and M4) produced only 53.6 L/lb. and 13.2 L/lb. of gas, respectively, in 233 days. With regular neutralization and recirculation of leachate, the rest of the food waste reactors ultimately overcame the very long lag phase, Reactor F7 with horse manure started producing gas again from day 232. Another horse manure reactor, F8, was dismantled on day 70 because it had not produced gas for a long time. The rest of the reactors were monitored for 579 days and then were dismantled. The highest amount of gas was produced by F4 (72 L/lb.), followed by F3 (68.5 L/lb.) and F7 (57.8 L/lb.). Food waste reactors with pig manure (F5 and F6) did not perform as well as MSW reactors (M5 and M6) in terms of gas production.

Figure 4-17 shows the cumulative methane generation in food waste reactors (F1 to F8). As mentioned in Section 4.4.2, the methane content was high in almost all of the food waste reactors. The highest methane volume was generated from reactor F4 (43.3 L/lb.), which was close to the volume of methane of reactor M6 with pig manure. The lowest amount of methane was produced from food waste reactor F5 (only 12.7 L/lb.)

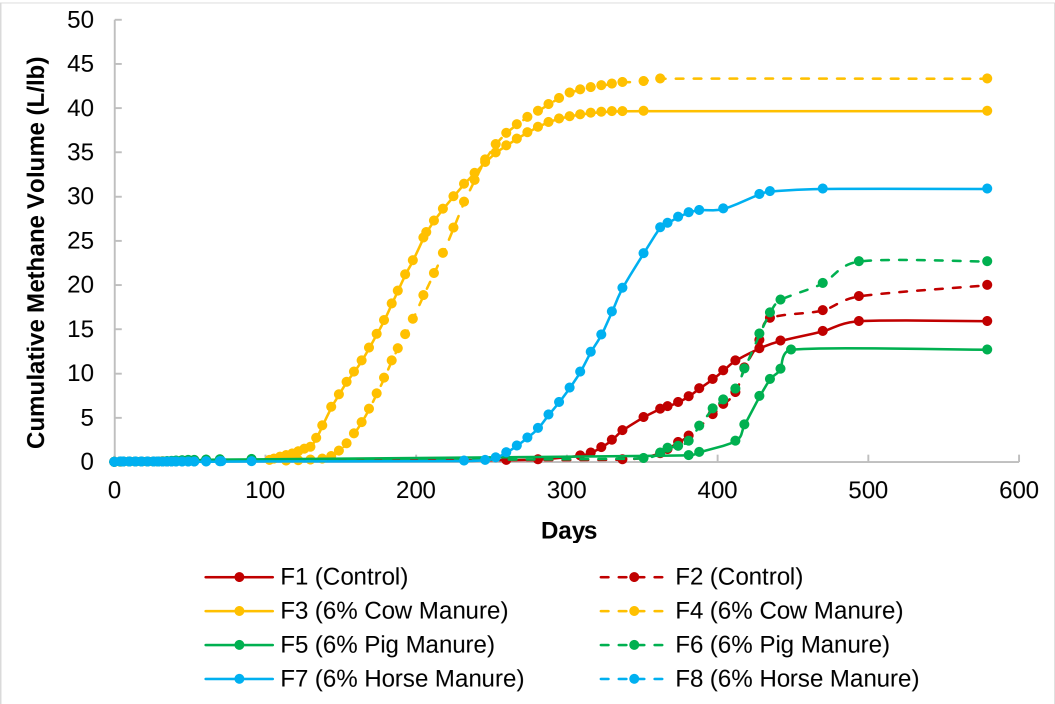


Figure 4-17 Cumulative methane generation (L/lb.) in food waste reactors

The control food waste reactors (F1 and F2) also produced significant amounts of methane (15.9 L/lb. and 20 L/lb., respectively), while the control MSW reactors produced very small amounts of methane (only 2.5 L/lb. and 8.2 L/lb.) In conclusion, food waste reactors were able to generate substantial amounts of methane, but it required more than twice the amount of time of MSW reactors.

In this study, the moisture content of food waste was more than 70% (Table 4-4). This may negatively affect the biocell operation, as the optimum moisture content for bioreactor operation is 40-55%, and more water then optimal in the waste might affect the methane yield. The volatile solid content of food waste was 91-93%, which indicates that the feedstock had high potential for biodegradability (Table 4-5). Karanjekar (2012) found that methane production for 100% food waste is quite low compared to other wastes and that production peaked after 160 days of operation, which could be due to the enhanced lag phase due to rapid hydrolysis. The longer lag phases can be attributed to volatile fatty acid (VFA) generation in reactors with high food content. Rapid hydrolysis and volatile fatty acid accumulation in waste with a high percentage of food waste cause an increased lag phase before methanogenesis started in Shao's et al. 2005 study. At least one study showed the effect of feed inoculum ratios on biogas yields, i.e. more inoculum in feedstock produces more biogas. Liu et al. (2009) showed that a 38% sludge addition has the highest methane yield. Various substances and conditions may cause inhibitory effects on the anaerobic digestion process. Anaerobic microbes require specific physical conditions to maintain enzyme activities to facilitate the biochemical reactions. Unfavorable conditions in anaerobic reactors, such as temporal overloading, a decreasing pH, and rapid temperature changes, inhibit anaerobic processes (Gallert et al., 1998). Apart from these factors, ammonia and long chain fatty acids (LCFA) also inhibit the anaerobic digestion process. In anaerobic digestion, ammonia, mostly in the form of protein, is primarily produced by the degradation of the nitrogenous matter present in the feedstock (Kayhanian et al., 1999; Kotsyurbenko et al., 2004). Ammonia is inhibitory to methanogenesis if it exists at high concentrations (Gallert et al., 1998), but concentrations between 50 and 200 mg L-1 have a beneficial effect for bacterial growth McCarty (1964). Ammonia, as a base, also neutralizes the volatile fatty acids produced by fermentative bacteria, and thus helps maintain the neutral pH conditions essential for cell growth (Jiang, 2012).

Figure 4-18 and Figure 4-19 show the gas generation rate and methane yield of the food waste reactors, respectively. From the gas yield versus time graph, the lag phases in food waste reactors can be observed clearly.

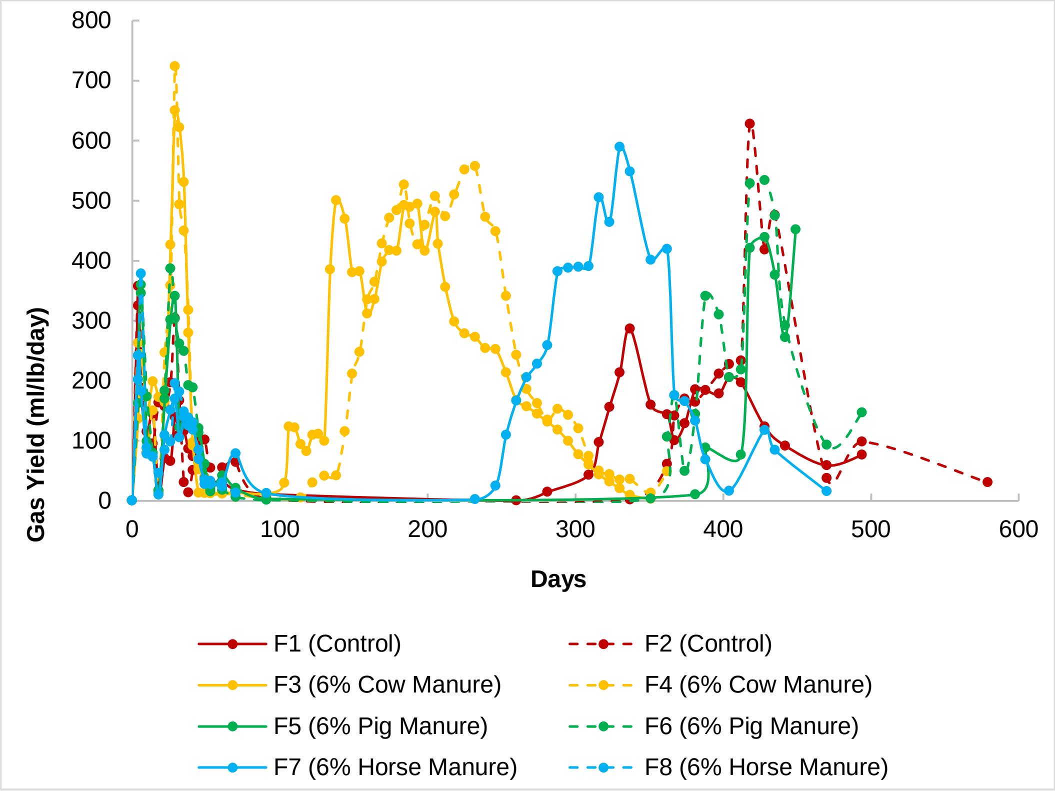


Figure 4-18 Gas yield (mL/lb./day) in food waste reactors

Initially, there was a substantial amount of gas production in food waste reactors up to 50 days. The gas generation rate in reactor F4 reached its peak of 723.7 mL/lb./day in only 29 days of operation, which is almost twice the rate of the rest of the reactors. Soon, all the reactors entered the lag phase, and the gas yield dropped significantly. Food waste reactors F3 and F4 again were in an increasing trend of yielding gas after 91 days. Both of these reactors displayed multiple peaks, but never reached the initial peak of 723.7 mL/lb./day. Reactor F7 reached its peak of 589.4 mL/lb./day on day 330; F5 and F6 reached their peaks (451.9 mL/lb./day) on day 449 day 428 and (533.9 mL/lb./day).

The methane yield versus time graph shows the methane generation rate with time (Figure 4-19), and shows that methane and gas yields in food waste reactors follow the same trend. Food waste reactors with cow manure (F3 and F4) experienced the methanogenic phase earliest, with the decomposition of waste taking place from day 100 to day 350. One of the horse manure reactors (F7) experienced methanogenic phase from day 246 to day 470; the other one (F8) never reached the methanogenic phase, as it was dismantled on day 70. Food waste control reactors (F1 and F2) and pig manure reactors (F5 and F6) reached the methanogenic phase around day 300 and kept producing methane until around day 494. When the reactors reached the methanogenic phase, the generation of methane peaked. Due to the heterogeneous properties of waste, the methane yields varied, even for the same pair of reactors. For example, reactor F3 with cow manure had the peak methane yield on day 184 (391.8 mL/lb./day), but another reactor with cow manure (F4) peaked on day 232 (415.2 mL/lb./day). F2, one of the control reactors, had the highest methane generation rate all of the food waste reactors, with a generation rate of 465.1 mL/lb./day on day 418.

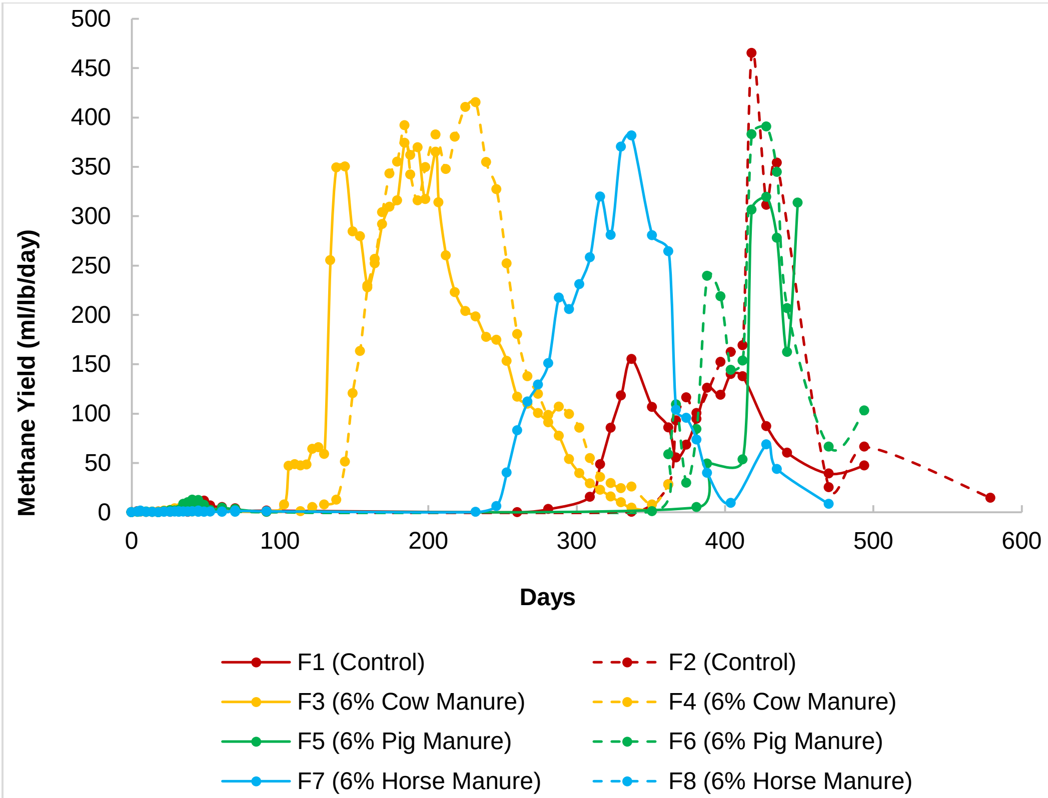


Figure 4-19 Methane yield (mL/lb./day) in food waste reactors

4.5 Leachate Characteristics

Leachate is generated by excess water percolating through the waste layers in a landfill. The chemical and biological processes of the waste are significantly influenced by the characteristics of the generated leachate such as pH, volume, BOD (biochemical oxygen demand) and COD (Chemical oxygen demand). The characteristics of generated leachate indicate the level of degradation of the solid waste. In this study, the pH, volume of generated leachate, BOD, and COD were monitored for both MSW and food waste reactors during their decomposition phases. Details are discussed in the following sections.

4.5.1 pH of Leachate of MSW Reactors

In the MSW reactors (M1 to M10), the pH of leachate was initially low (less than 6) in most of the reactors, indicating the acidity of the MSW. In the initial phase of decomposition of MSW, carbohydrates, lipids, and proteins are first broken down into soluble monomers, followed by the formation of volatile fatty acids (lactic, propionic, and butyric acids, etc.). In these phases, the pH drops significantly due to the presence of acid. Excessive accumulation of acid impedes the growth of methanogenic bacteria. To have the maximum methane production, the pH should be within a range of 6.6 to 7.5 (Metcalf and Eddy, 2004). A pH of 6.8 is optimal and has a minimum lag period for methane production (Lay et al., 1997). If the pH drops below 6.6, methanogenic bacteria are significantly inhibited; pH below 6.2 is toxic (Metcalf and Eddy, 2004). In this study, when the pH dropped below 7, potassium hydroxide (KOH) was added with the collected leachate to neutralize the pH, and the neutralized leachate was recirculated in the reactors. This maintained a favorable environment for bacterial growth inside the reactors. The pH began increasing because of the frequent neutralization and recirculation of leachate, and the transition phase, between acidogenic phase and the methanogenic phase, started when the methanogens became active. Methanogens are strictly anaerobic and converted the acetic acid/acetate to methane and carbon dioxide in the methanogenic phase. The pH value increased to 7. The pH in all of the MSW reactors was higher than 6 within 50 days and reached or exceeded 7 within 110 days of operation. The variations of pH with time in the MSW reactors (M1 to M10) are shown in Figure 4-20. Leachate with a pH of more than 5 was produced for the first time on day 4 of operation in the MSW reactors (M1 – M10). On day 8, the pH dropped significantly in all the reactors. In the MSW control reactors (M1 and M2), the pH dropped from 5.23 to 4.29 in M1 and from 6.29 to 5.2 in M2.

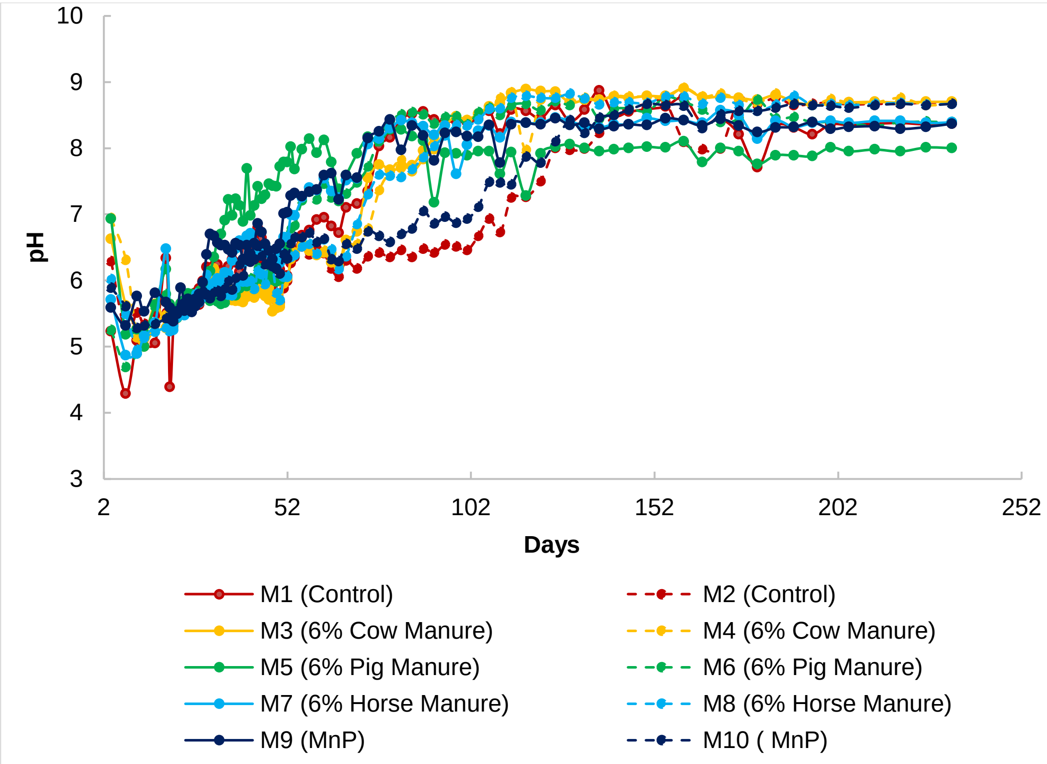


Figure 4-20 pH of leachate of MSW reactors

The pH began gradually increasing in both of the control reactors and reached 6 on day 30 in M1 and on day 43 in M2. The leachate's pH in reactors M1 and M2 reached 7.1 on day 68 and 7.25 on day 113, indicating the methanogenic phase. The pH remained at more than 7 and stabilized at maximum values of 8.87 and 8.79 for M1 and M2, respectively. In the MSW reactors with cow manure, M3 and M4, the initial pH was 6.63 and 6.94, respectively, on day 4. Following a trend similar to that of M1 and M2, the pH of M3 and M4 dropped on the next monitoring day (day 8), and kept decreasing. The lowest value of leachate pH was observed on the 13th day of operation (5.11 and 5 for M3 and M4, respectively). The pH of leachate in M3 and M4 was more than 6 on day 52, and entered the methanogenic phase on days 74 and 77, respectively. M3 and M4 had the highest values of pH (8.9 and 8.91, respectively) on day 160. MSW reactors with pig manure (M5 and M6) had an initial pH of 6.93 and 5.25, respectively, on day 4. On day 8, the pH dropped to 5.18 and 4.69 for M5 and M6, respectively, then began increasing. The pH of leachate of M5 and M6 reached to more than 6 on day 31 and day 42, respectively, and methanogenic phase started on days 38 and 56, respectively. M5 and M6 had the highest values of pH (8.28 on day 80 and 8.77 on day 155, respectively). For the MSW reactors with horse manure (M7 and M8), the initial pH was 5.71 and 6.02, respectively, on day 4. As in the other MSW reactors, the lowest values of pH were observed (4.87 and 4.97) for reactors M7 and M8, respectively, on day 8. The pH of leachate in M7 and M8 reached to more than 6 on day 35 and day 44, and the methanogenic phase started on day 56 and day 74, respectively. The maximum pH for M7 and M8 was measured as 8.57 and 8.88, respectively, on day 170 and day 190. M9 and M10, with the MnP enzyme, produced the highest amount of methane of all of the MSW reactors. The pH on day 4 was 5.59 in M9 and gradually decreased on day 8; the pH was 5.89 in M10 on day 4 and gradually decreased on day 11. In both of the reactors, the pH started increasing and reached to more than 6 on day 31 for M9 and on day 38 for M10. On day 51, the pH of M9 reached more than 7; on day 85, the pH of M10 reached more than 7. The highest pH values measured for M9 and M10 were 8.45 and 8.71, respectively. From the pH versus time plot of MSW reactors and methane generation graph, it can be seen that MSW reactors with pig manure and MnP reached the methanogenic phase earlier than other MSW reactors.

4.5.2 pH of Leachate of Food Waste Reactors

A significant drop in the pH in food waste reactors (F1 to F8) was observed throughout the initial monitoring period due to excessive volatile fatty acid (VFA) accumulation in food waste. Previous researchers (Shao et. al., 2005; Karanjekar, 2013) also experienced a pH drop in food waste due to VFA accumulation. KOH was added with the leachate during recirculation to neutralize the pH. From the pH vs time plot of food waste reactors (Figure 4-21) it was noticed that the initial pH was less than 5 for as long as 10 days, which may have retarded the bacterial growth in the reactor. After the addition of 10% sludge at 20 days, the pH increased, but not significantly. After 45 days of operation, the pH of most of the reactors reached or surpassed 6, except for the horse manure reactors (B7 and F8) which had minimal methane production (Figure 4-17). In comparison, it took about 110 days for the MSW reactors to get to the methanogenic phase (Figure 4-20), but only reactor B3 of the food waste reactors was able to achieve more than 7 after 110 days; it began producing methane earlier than other food waste reactors. It took almost 370 days for all of the food waste reactors to attain the methanogenic phase (Figure 4-21).

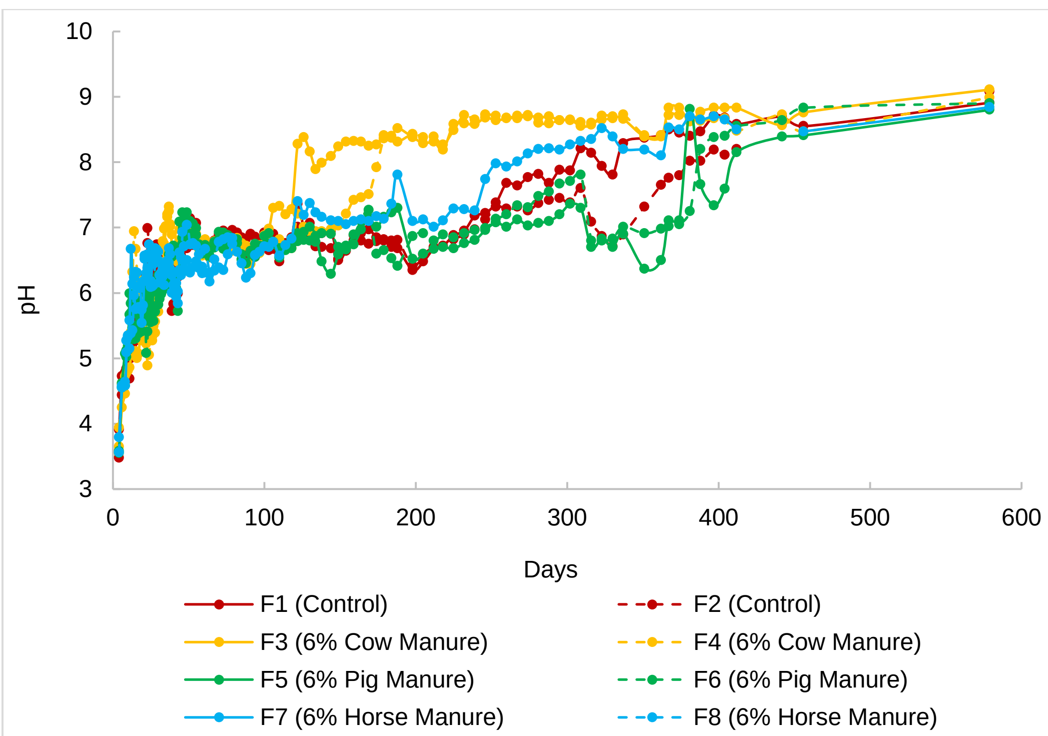


Figure 4-21 pH of leachate of food waste reactors

Leachate was produced for the first time, and the pH (3.48 to 3.91) was the lowest on day 4 of 579 days of operation of the food waste reactors In the food waste control reactors (F1 and F2), the pH was also the lowest (3.91 and 3.48) on day 4. The pH gradually began increasing in both of the control reactors, after the addition of sludge on day 20, reached 6 on day 21. The pH of the leachate of reactors F1 and F2 was 7.18 on day 239 and 7.12 on day 246, respectively, which indicated the methanogenic phase. In comparison, the MSW control reactors reached the methanogenic phase after only 68 and 113 days. Once the reactors reached the methanogenic phase, the pH remained at more than 7 and then stabilized with maximum values of 8.91 and 9.08 for F1 and F2, respectively.

In the food waste reactors with cow manure (F3 and F4), the initial pH was 3.65 and 3.94, respectively, on day 4. Following a trend similar to F1 and F2, the pH of F3 and F4 was found to have increased on the next monitoring day (day 8), and kept climbing. The pH of leachate of F3 and F4 was more than 6 on day 30, and entered the methanogenic phase on days 106 and 122, respectively. F3 and F4 had the highest values of pH, 9.11 and 8.99, respectively, on day 579. Food waste reactors with pig manure (F5 and F6) had an initial pH of 3.55 and 3.58, respectively, on day 4. From day 8, the pH kept rising, with F5 attaining more than 6 on day 27 and F6 attaining more than 6 on day 33. The pH of F5 and F6 was higher than 7 on day 260, with frequent fluctuations, and stabilized with a maximum pH of 8.9 and 8.84, respectively, on day 579. For the food waste reactors with horse manure (F7 and F8), the initial pH was found to be 3.79 and 3.56, respectively, on day 4. The pH of the leachate in F7 and F8 was 6 on day 21 and day 14, respectively, and the methanogenic phase started on day 122 in F7. B8 was dismantled on day 70, before entering the methanogenic phase, because it had not produced any gas. The maximum pH values for B7 and B8 were 8.84 and 6.848, respectively, on days 579 and 70.

In this study, the low pH in the initial stage had an overall effect on methane production. In a study conducted by Wang et al. (1997), an initial pH of 3.4 to 3.7 in 70% food waste and 30% old refuse reactors caused high accumulations of VFA and ammonia, which led to the termination, on day 149, of reactors that had produced very little methane. Despite pH neutralization by sodium carbonate, these reactors failed to undergo methanogenesis. The accumulation of VFAs and the high COD indicated that the conversion of soluble organic carbon by syntrophic activity of acetogenic and methanogenic bacteria limited methane production over the 149-day period (Wang et al., 1997).

4.5.3 Chemical Oxygen Demand (COD) of MSW Reactors

COD of the leachate in MSW reactors was measured on a monthly basis to determine the level of degradation of waste inside the reactors (Figure 4-22). Initially, COD in all of the reactors had high values, which indicated the commencement of the anaerobic phase, when there is a deficiency of oxygen for the microbes in the leachate. Due to the lack of oxygen and transition to the anaerobic phase, the COD concentration increased, as the hydrolysis continued. The COD concentration kept rising until the methanogenic phase began, and then dropped with the increase in activity of methanogenic bacteria, which was indicated by the increase of the methane generation rate. According to Alkaabi et al. (2009), the COD value decreases initially, due to the aerobic phase, and then increases. According to Wang et al. (1997), when the reactor reaches the methanogenic phase, the COD drops significantly. Other than MSW reactors M5 and M9, all of the reactors followed the same trend from the beginning. The first test was conducted one month after the reactor was setup. In reactors M5 and M9, the COD was very low from the beginning and was a descending trend until the fifth month, which indicated that these reactors had been in the methanogenic phase since month one. In The COD of leachate in M5 was 24,668.77 mg/L in the first month and decreased to 6,711.84 mg/L by the end of the study, in the seventh month. The COD in M9 was 42,669.49 mg/L in the first month and decreased to 3,293.39 mg/L by the end of the study.

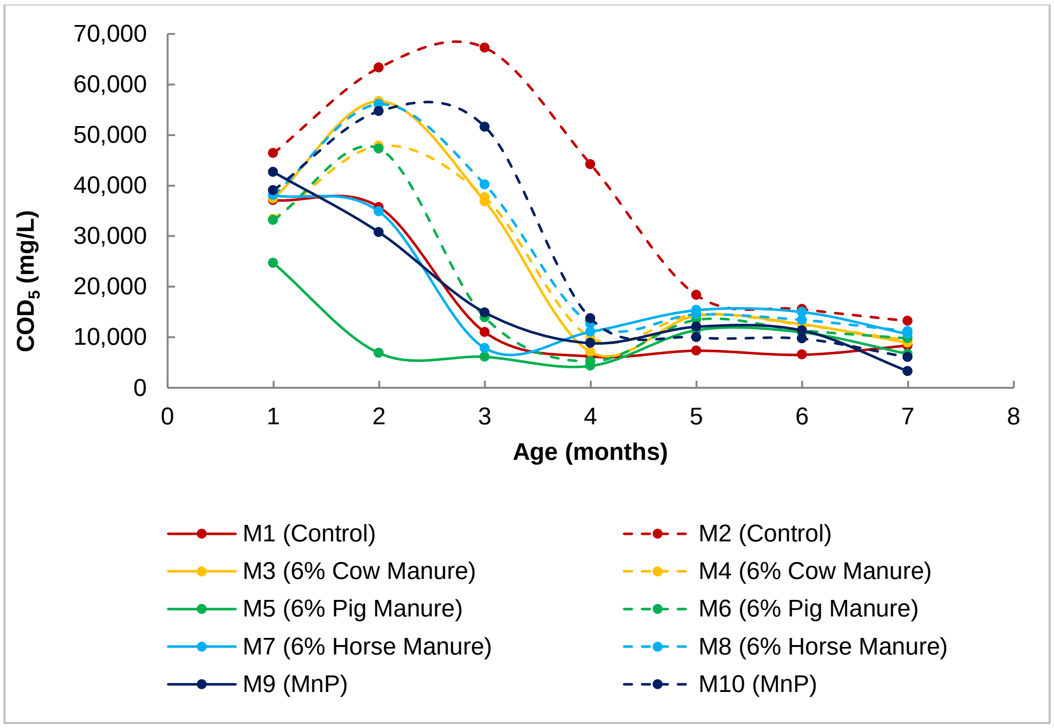


Figure 4-22 Chemical oxygen demand (COD) of leachate of MSW reactors

4.5.4 Chemical Oxygen Demand (COD) of Food Waste Reactors

The COD of the leachate of the food waste reactors was also measured on monthly basis to determine the level of degradation of waste inside the reactors, as shown in Figure 4-23. The COD value decreases with the degradation of waste. The initial COD values were high in all of the food waste reactors. Other than those with cow manure (F3 and F4), the COD values remained high in all the reactors due to a long lag phase. The COD of F3 and F4 started dropping the reactors started producing methane. The initial COD for reactors F3 and F4 were 146,385 mg/L and 138,559 mg/L, respectively, and decreased to 68,007.5 mg/L and 55,093.9 mg/L, respectively, at the end of nineteenth month.

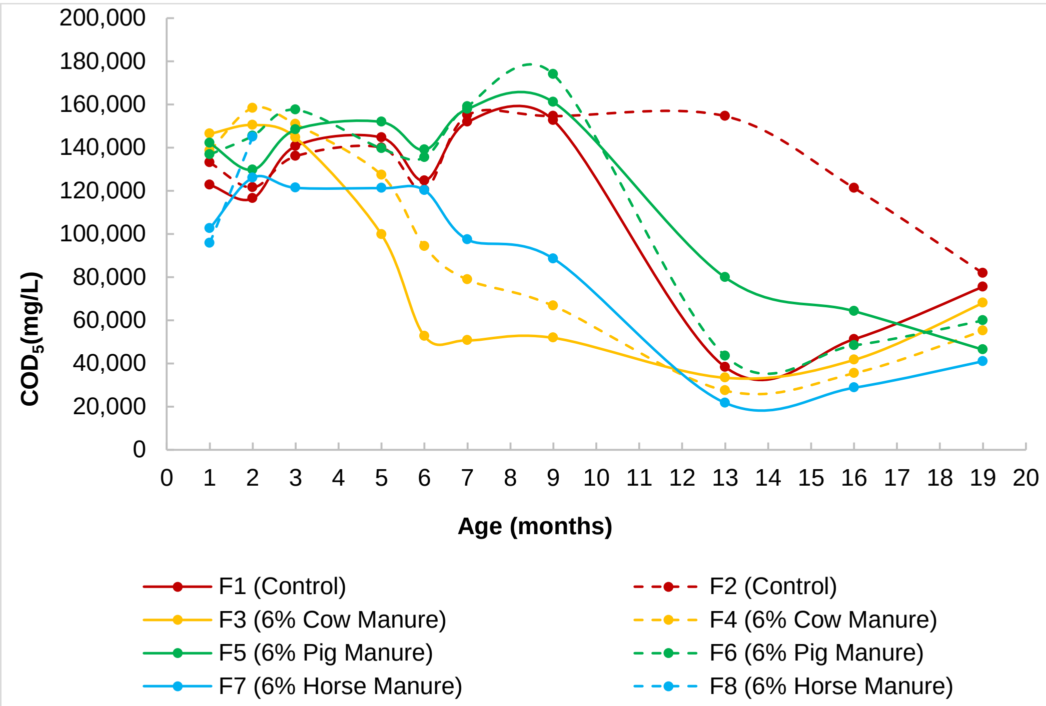


Figure 4-23 Chemical oxygen demand (COD) of leachate of food waste reactors

4.5.5 Biochemical Oxygen Demand (BOD) of MSW Reactors

The BOD of the leachate of the MSW reactors was also measured on monthly basis to determine the most beneficial microbial concentration to oxidize carbonaceous and nitrogenous compounds present in the leachate (Figure 4-24). The BOD values for the MSW reactors followed the same trend as COD values. From the beginning, a sharp decrease in BOD value was observed for reactors M5 and M9, showing that they were in the methanogenic phase even before the first BOD was measured at the end of the first month. The initial BOD values of M5 and M9 were 16,612.5 mg/L and 25,800 mg/L, respectively, and were reduced to 187.55 mg/L and 169.92 mg/L, respectively, at the end of the monitoring, in the seventh month.

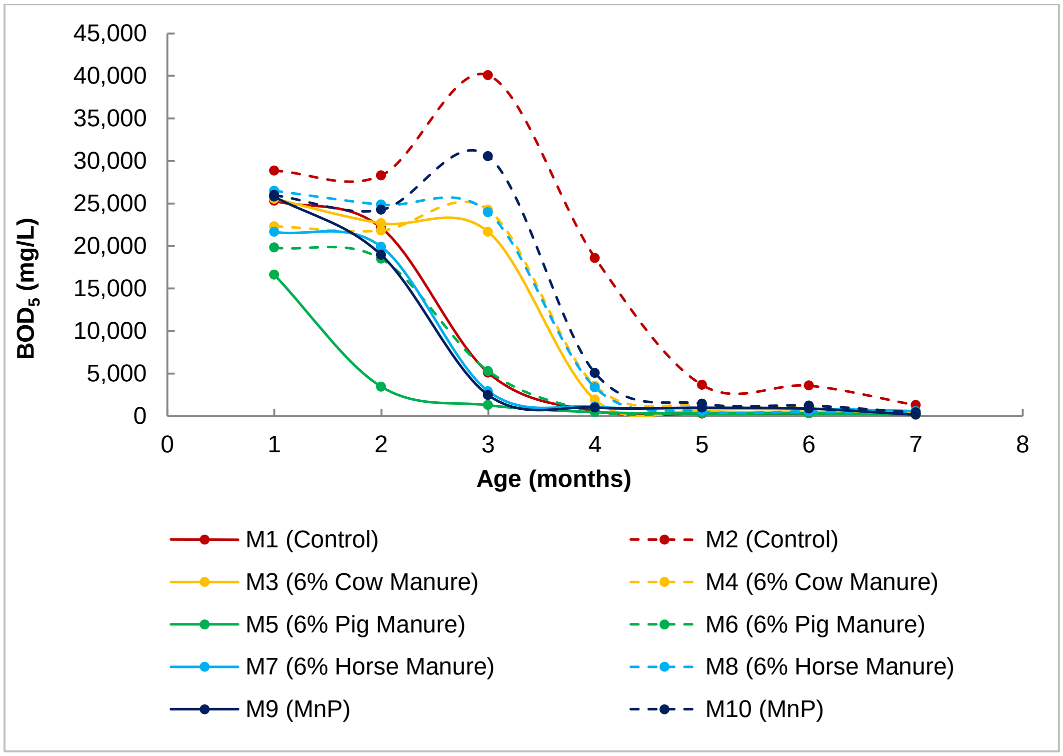


Figure 4-24 Biochemical oxygen demand (BOD) of MSW reactors

The BOD and COD ratios indicate the biodegradability of organic compounds present in the leachate, i.e. proportion of biologically degradable organic matter to total organic matter. This ratio decreases as the landfill ages and more degradation products are leached from deposited residues (Reinhart et al., 1998). According to Warith (2002), leachate is highly biodegradable when the BOD and COD ratio is about 0.4 to 0.8. Kjeldsen et al. (2002) reported that the acidic phase starts when the values of BOD/COD are around 0.58; in the methanogenic phase, the BOD/COD value is about 0.06.The BOD and COD ratios for all of the MSW reactors are shown in Figure 4-25.

At the beginning, the BOD/COD ratios for all the MSW reactors were about 0.7 to 0.6, which indicated the presence of biologically degradable organic matter in the feedstock. As the reactors started producing methane, this ratio was reduced, and at the end of the study, the ratio was at its lowest. During the second month of operation, the BOD/COD of all of the MSW reactors, except M1 and M9, fell below 0.58, indicating the start of the acidic phase, according to Kjeldsen et al. (2002). Within five months of operation, the BOD/COD of all of the MSW reactors, except M2 and M10, reduced to 0.06, reflecting the methanogenic phase. At the end of the study, after seven months of operation, all of the MSW reactors except M2 had BOD/COD of 0.06 or less, indicating the decomposition of waste was almost complete.

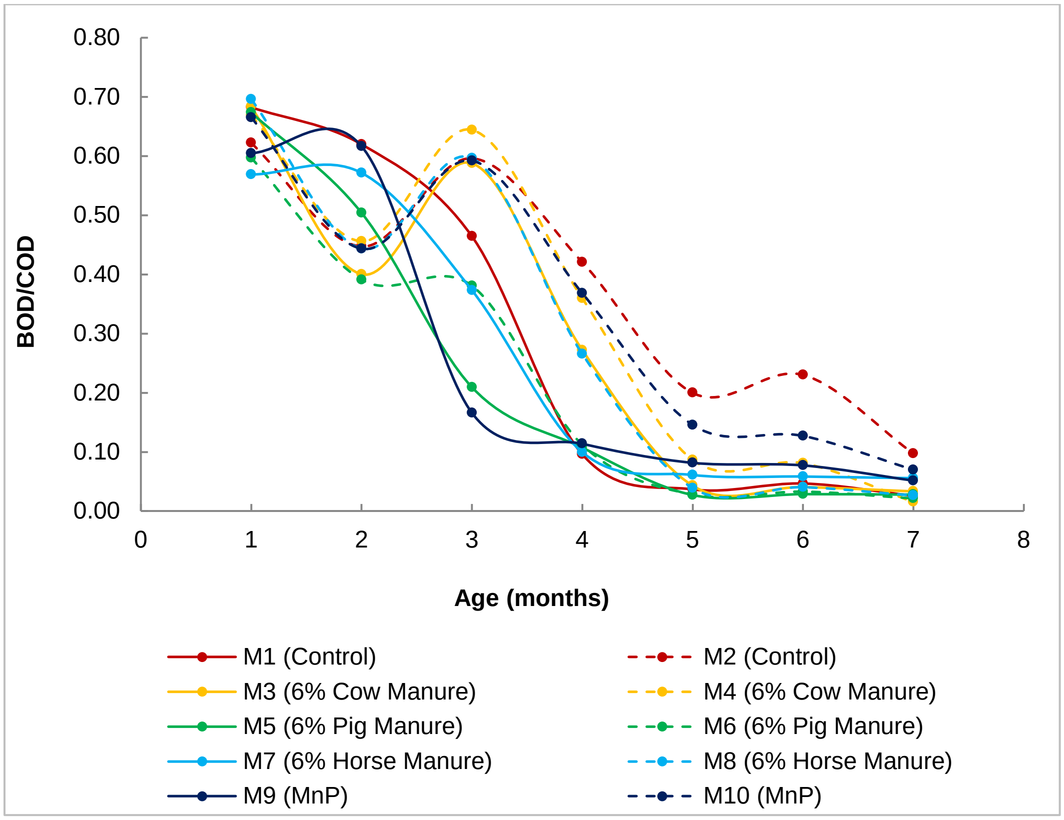


Figure 4-25 BOD/COD of leachate of MSW reactors

4.5.6 Biochemical Oxygen Demand (BOD) of Food Waste Reactors

The BOD of the leachate in the food waste reactors was measured during the 19 months of monitoring. (Figure 4-26). The initial BOD was as low as 71,280 mg/L for reactor F8 and as high as 101,025 mg/L for reactor F3. The BOD of all of the food waste reactors increased until the third month, which indicated the acidogenic phase, according to Barlaz et al. (1993). It started dropping in the fourth month of operation, which indicated decomposition of biological materials in the waste. At the end of the operation, after 18 months, the BOD value was lowest in most of the reactors, ranging from about 6,233 mg/L to 27,011 mg/L.

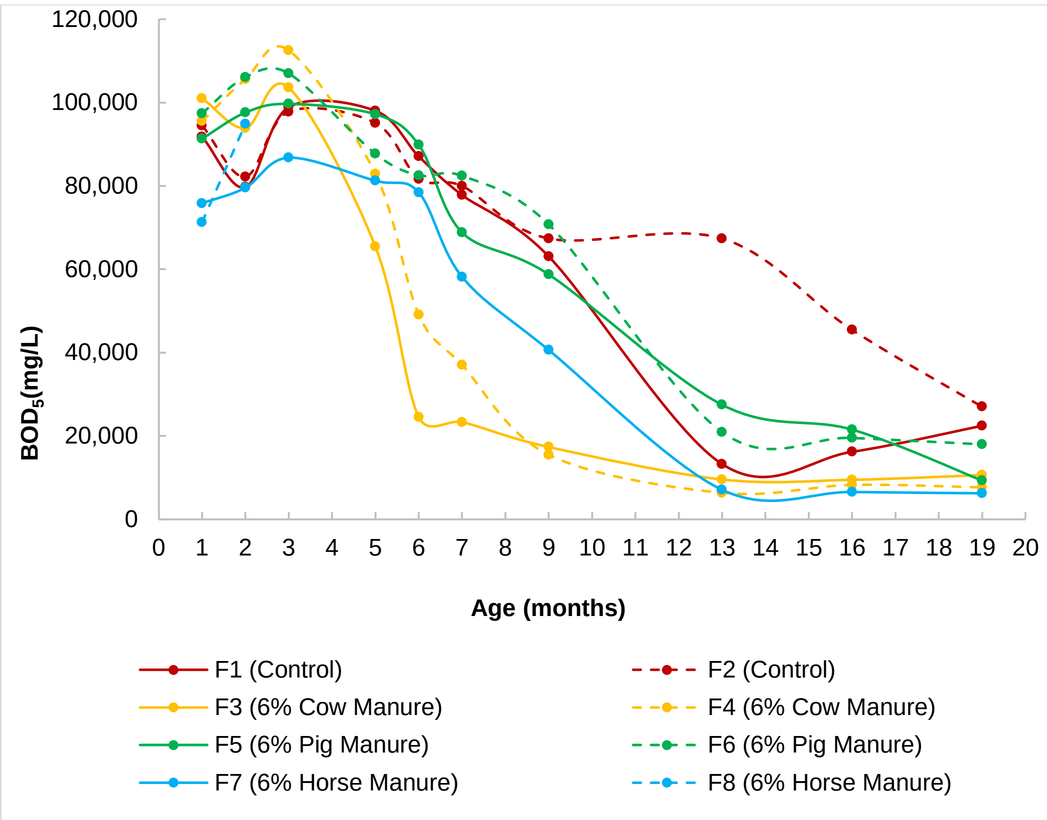


Figure 4-26 Biochemical oxygen demand (BOD) of leachate of food waste reactors

Figure 4-27 depicts the BOD/COD of leachate for all of the food waste reactors. Initially, the BOD/COD ratios were about 0.75 to 0.64, which indicated the presence of biologically degradable organic matter in the feedstock. As the reactors started producing methane, this ratio decreased. Similar to the MSW reactors, at the end of the study, the ratio was the lowest. The BOD/COD fluctuated rapidly up until the fifth month and the value was always more than 0.6, except for the reactors with cow manure (F3 and F4). Only F3 and F4 were able to overcome long the acidogenic phase and begin producing methane. Even after 19 months of the study, none of the food waste reactors were able to get BOD/COD of less than 0.1. The lowest values of BOD/COD, about 0.14 to 0.33, were measured at the end of the study, after 19 months, indicating the presence of degradable material.

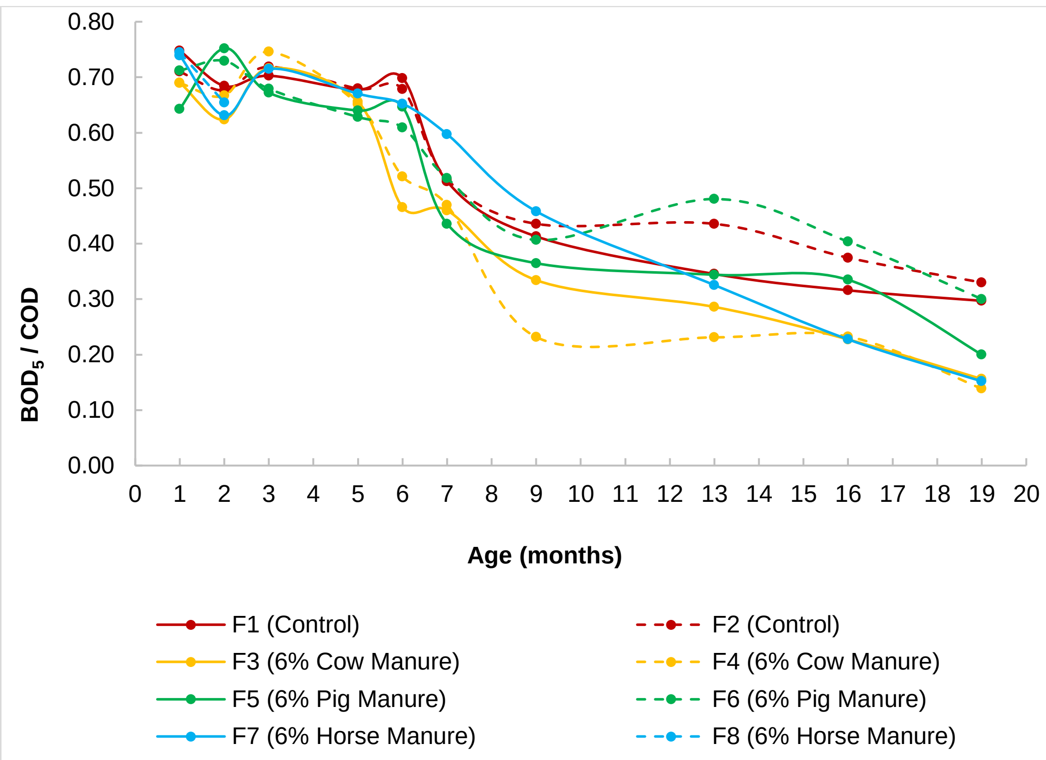


Figure 4-27 BOD/COD of leachate of food waste reactors

4.6 Degree of Waste Stabilization

4.6.1 Weight Loss and Settlement

Waste stabilization can be determined by evaluating the leachate's ratio of BOD/COD and the volatile solid content in the waste after decomposition. In this study, both MSW and food waste reactors were monitored as long as they produced methane. The MSW reactors started producing methane and stabilized earlier than the food waste reactors. After 241 days of operation, the 10 reactors filled with MSW (M1 to M10) were dismantled, and the characteristics of their waste characteristics was measured in terms of weight loss, total settlement, moisture content, volatile organic content, etc. Figure 4-28 shows the percentage of weight loss and settlement after decomposition of MSW in reactors M1 to M10.

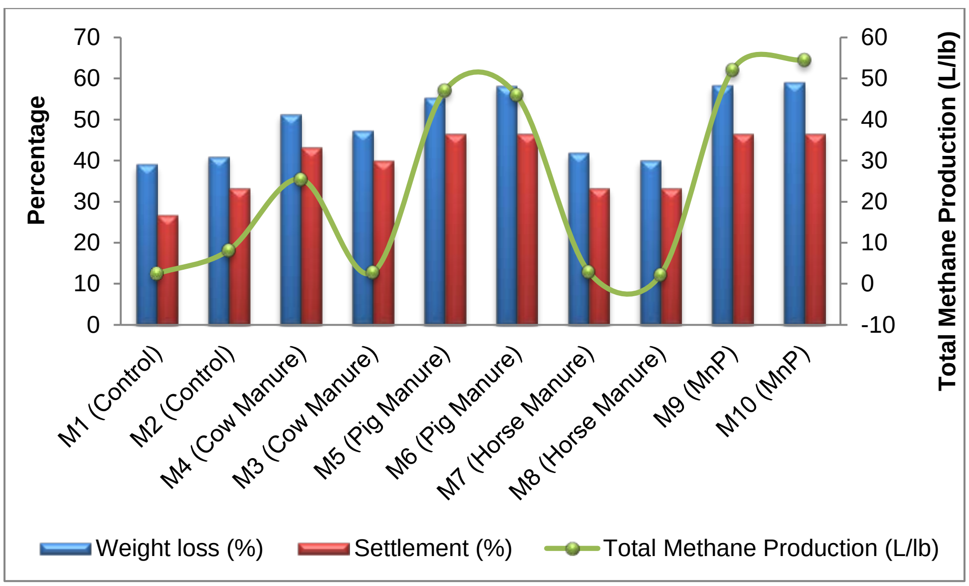


Figure 4-28 Percentage of weight loss and settlement after decomposition of MSW

Reactors with MnP, i.e. reactors M9 and M10, experienced the highest percentage of weight loss, which was about 58.32% and 59.01%, respectively. Reactors with pig manure (M5 and M6) had weight losses of about 55.25% and 58.28%, respectively. The control reactors had the lowest amount of weight loss (about 39.02% and 41.07%, respectively). A change in the height of waste in the reactors indicates settlement of the waste, which is proportional to its weight reduction. Initially the height of waste in all of the MSW reactors was about 6 inches. After the decomposition of waste, the height change was different for different reactors, according to the total methane production. As the reactors M5, M6, M9 and M10 had the highest methane production; wastes in these reactors settled the most and lost the most weight.

After 579 days of operation, reactors filled with food waste (F1 to F7) were dismantled, and waste characteristics were measured in terms of weight loss, total settlement, moisture content, volatile organic content, etc. Figure 4-29 shows the weight loss and settlement in food waste reactors after degradation.

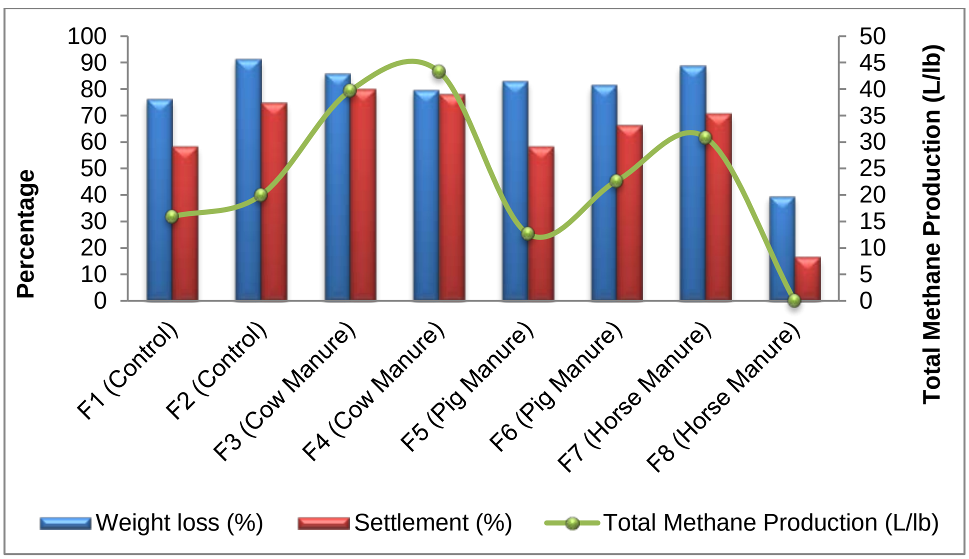


Figure 4-29 Percentage of weight loss and settlement after decomposition of food waste

The lowest amount of waste loss was observed in reactor F8, as it was dismantled after only 70 days of operation. After a lag phase, the other reactors started producing gas and were monitored for 579 days. When reactors F1 to F7 did not produce gas for a long time, destructive tests were performed. The highest amount of waste loss was seen in reactor F2 (91.43%), followed by reactor F7 (88.99%). Settlement in the reactor is proportional to gas production, i.e., where gas production is more, the settlement of waste is higher. The highest settlement was seen in reactor F3 (80%), followed by F4 (78%); gas productions were the highest in F3 and F4 (39.66 L/lb. and 43.32 L/lb., respectively). Unlike the MSW reactors, pig manure did not perform well in terms of gas production in food waste reactors, and produced 12.69 lb./L and 22.66 lb./L methane.

The moisture content of the feedstock of each MSW and food waste reactor was determined before filling the reactor and again at the end of the study, after dismantling, to determine the change in moisture content (Figure 4-30 and 4-31).

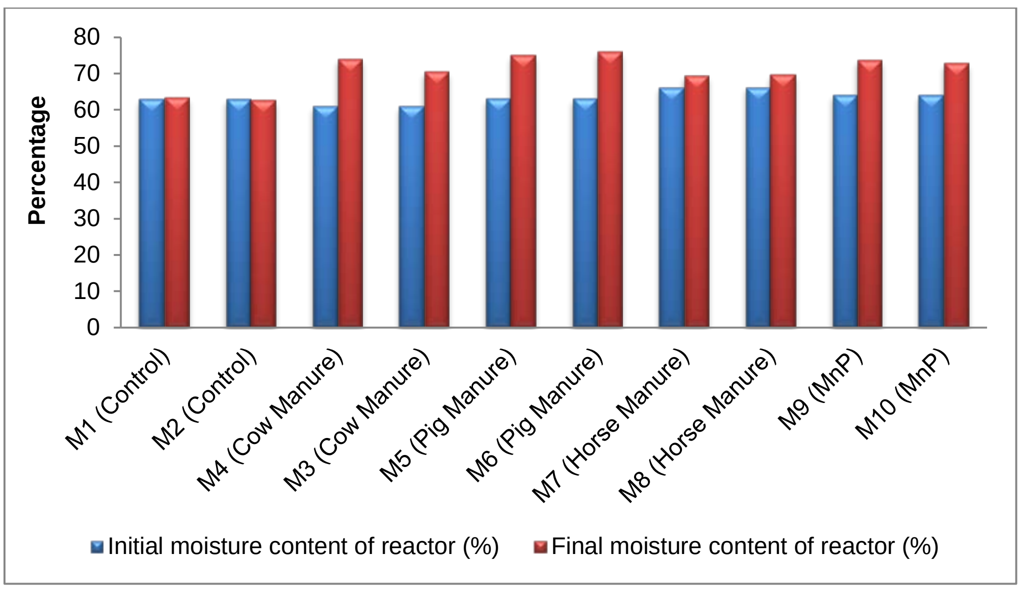


Figure 4-30 Change in moisture content after degradation of MSW

Though the initial moisture contents were similar in all the reactors, due to different levels of degradation, the final moisture content varied. The final moisture content of the MSW feedstock was higher than the initial moisture content. According to Haque (2007), the average moisture content of MSW increases with decomposition, as degraded MSW particles retain more moisture due to the decrease in pore spaces. The highest moisture content was found in in reactors M5 and M6 (75.05% and 75.97%, respectively). Each pair of MSW reactors with same inoculum seemed to have similar moisture contents.

The moisture content of most of the degraded food waste samples at the end of the study was found be greater than the initial moisture content, except for reactors F3, F4 and F5. The lowest moisture content was observed in reactor F4, followed by F3. These two reactors produced the highest amount of methane, which indicates that the moisture in the reactors was used up to break down the food waste. Reactor F2 had the highest moisture content (80.49%) and produced almost twice the amount of methane as reactors F3 and F4. Figure 4-31 shows the changes in moisture content in the food waste reactors (F1 to F8).

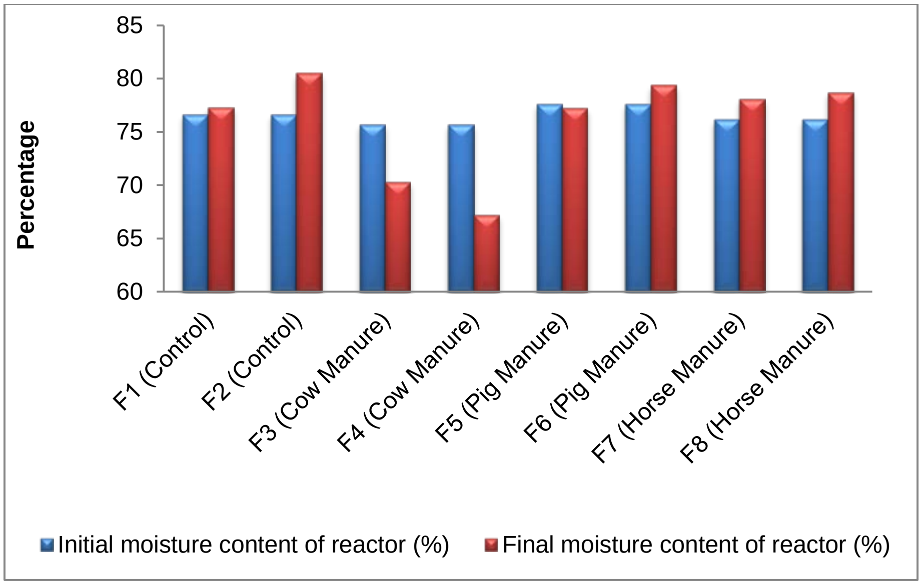


Figure 4-31 Change in moisture content after degradation of food waste

4.6.2 Reduction of Volatile Solid Content

The volatile organic content of degraded waste in the reactors was measured at the end of the study to investigate the effect of manure and enzymes on the degradation of MSW and food waste. The amount of final volatile solids in the degraded MSW and food waste were compared to the initial values of the fresh waste of the reactors to measure the percentage of degraded volatile solids at the end of the laboratory simulation of a biocell. It was concluded that the percent of the reduction of volatile solids in the waste is positively related to the total methane production from the waste, i.e. the more volatile solid reduction, the more methane generation. Figure 4-32 shows the changes in the volatile solid content of the feedstock of MSW reactors (M1 to M10) after decomposition.

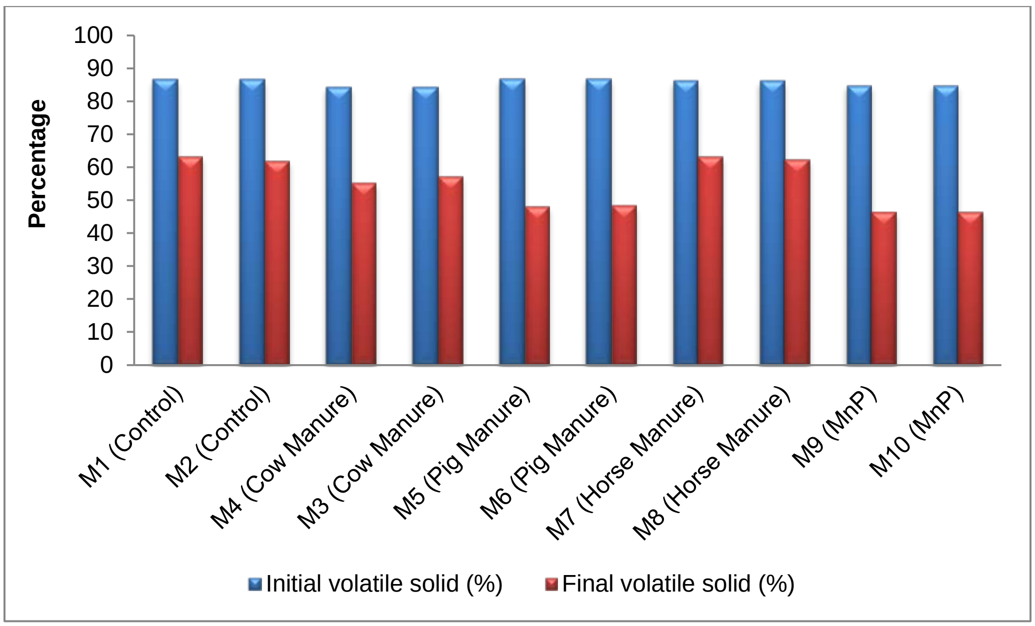


Figure 4-32 Change in volatile solid after degradation of MSW

The maximum amount of volatile solids (77.01% and 77.27%) was removed from reactors M9 and M10, respectively, as shown in Figure 4-33. These two reactors produced the most methane (52.03 L/lb. and 54.50 L/lb., respectively). The reactors with pig manure (M5 and M6) had the second highest production of methane, which was also noticed from the depletion of volatile solids in these reactors. When the total amount of methane generation from MSW reactors was plotted against the percent reduction of volatile solid of the waste, it rendered a linear trend of R^2 of 0.9304, which showed that the methane production from waste can be correlated with the reduction of degradation of volatile solids in the waste. Figure 4-34 shows the relationship between volatile solid content and methane production in MSW reactors.

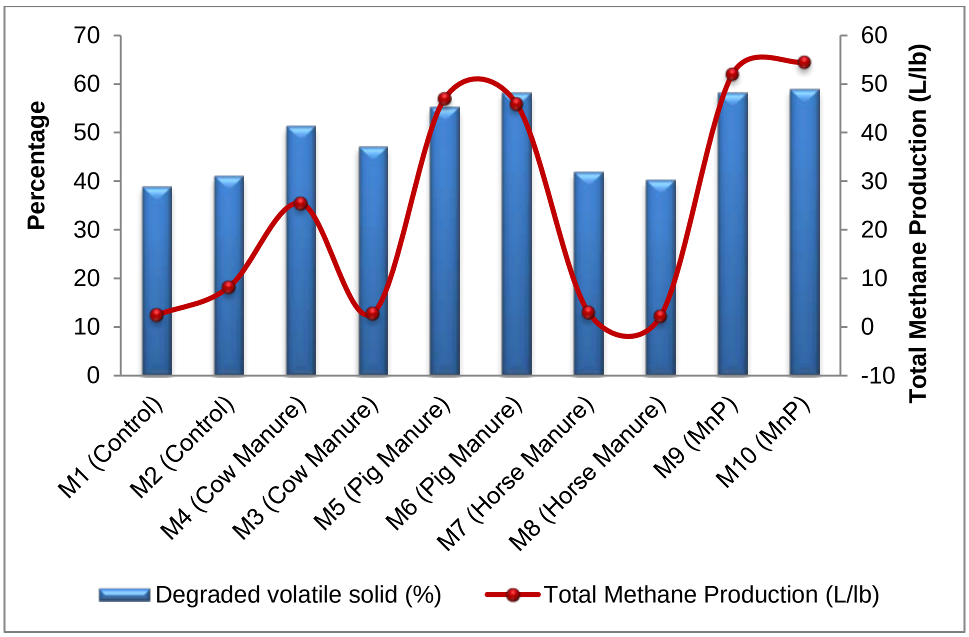


Figure 4-33 Volatile solid reduction of MSW after degradation

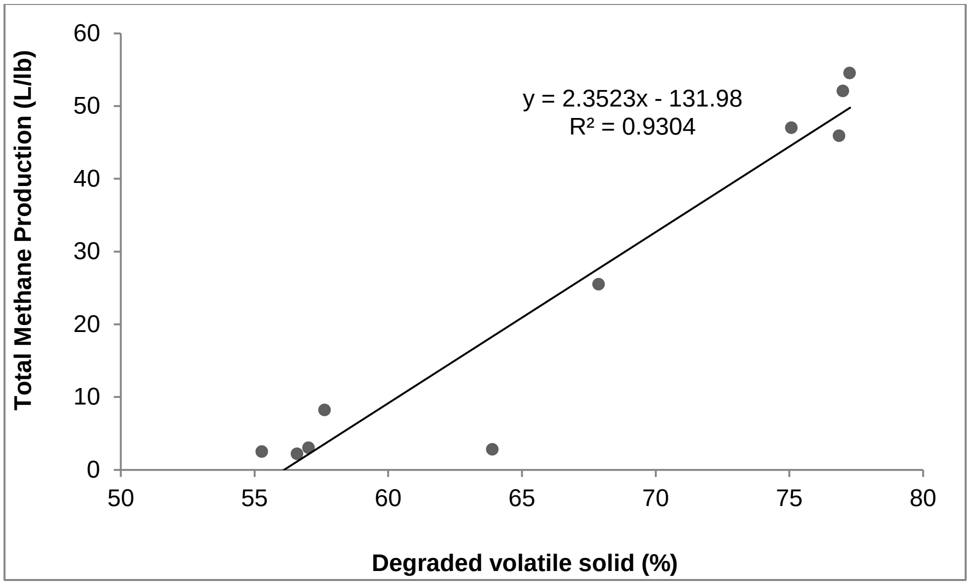


Figure 4-34 Relationship between volatile solid content reduction and methane production in MSW reactors

Changes in the volatile solid content were measured also for food waste reactors after 579 days, as shown in Figure 4-35. The final volatile solid content in the food waste reactors was lowest for reactors F3 and F4 with cow manure; they also produced the most methane. Haque (2007) observed that the final volatile organic content decreased to 46% from the initial volatile organic content of 91.5%. Sapkota (2017) reported that the reactor which produced the highest amount of methane experienced a reduction in solids to 46.63%. Al-Kaabi et al. (2009) reported that the reduction in volatile organic content was about 66% to 84% after anaerobic degradation with leachate recirculation. Sivanesan (2012) reported that the reduction in volatile solids was 84%, 82%, 77%, and 70% for the reactors with sludge. Figure 4-36 shows the percent reduction of volatile solid reduction of food waste after degradation, and Figure 4-37 shows the relationship between volatile solid content reduction and methane production in food waste reactors.

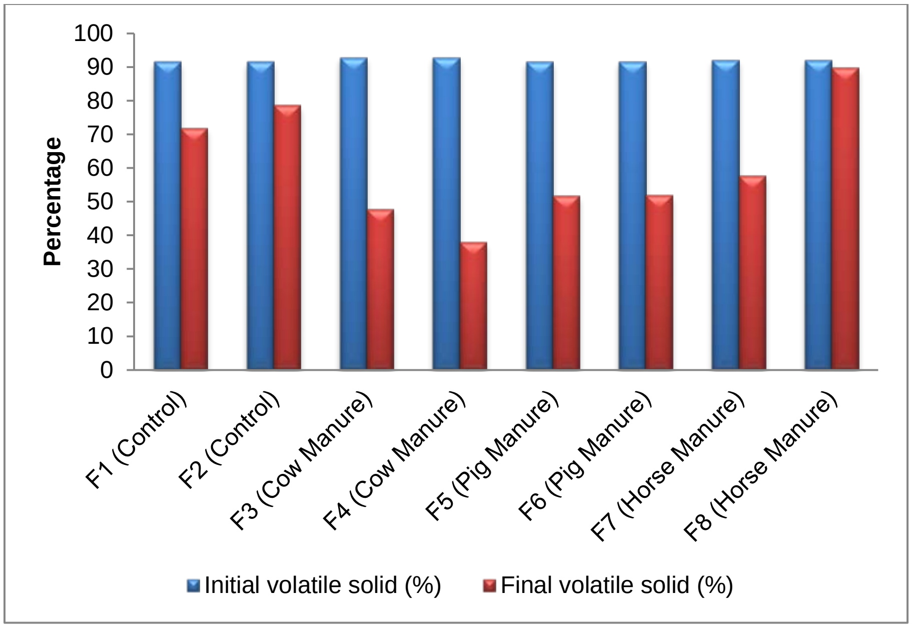


Figure 4-35 Change in volatile solid after degradation of food waste

Volatile solid removal is directly related to methane production, as depicted in Figure 4-36. The reactor which produced the highest amount of methane lost the most volatile solids during decomposition. The food waste reactor with horse manure, F8, which was dismantled during the monitoring on day 70, produced the least methane and decomposed the smallest amount of volatile solids.

To find the relationship between volatile solids and methane production, the total amount of methane generation from food waste reactors was plotted against the percent reduction of volatile solids of the waste. Unlike MSW, food waste has an exponential trend with R^2 of 0.9639, which shows that methane production from food waste is highly related to the reduction of degradation of volatile solids in the waste. In both MSW and food waste, the degree of waste stabilization can be explained from the weight loss, height settlement, moisture content, and volatile solid content of waste after the end of the laboratory scale biocell simulation.

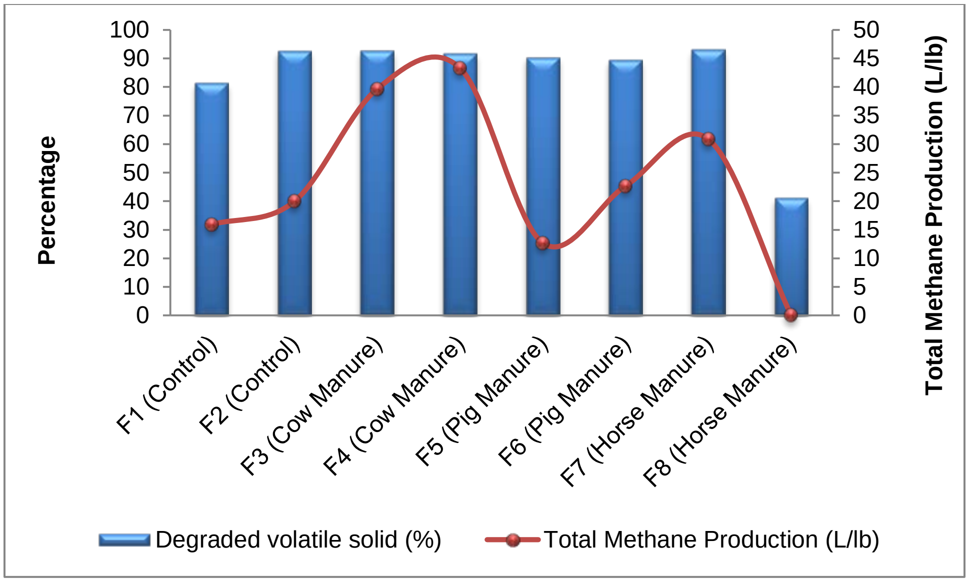


Figure 4-36 Volatile solid reduction of food waste after degradation

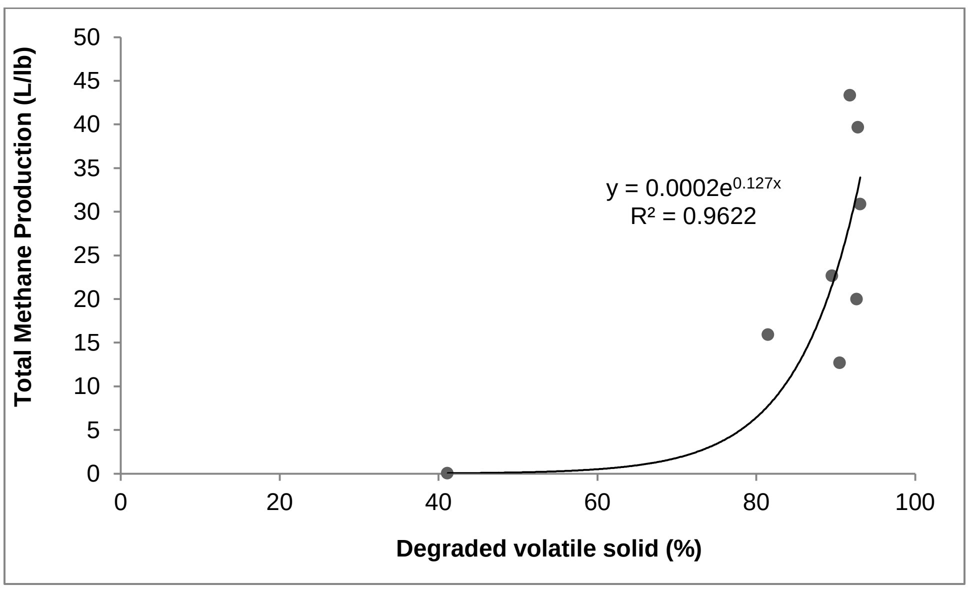


Figure 4-37 Relationship between volatile solid content reduction and methane production in food waste reactors

4.6.3 Reduction of COD and BOD of Leachate

At the end of the monitoring periods of MSW and food waste reactors, the final COD and BOD of all leachate samples were measured. Percent of peak reduction of COD and BOD values were calculated based on the peak and end concentrations during the anaerobic process. The percentages of BOD peak reduction values are similar for all of the MSW reactors, but the peak reduction values of the COD varied. The highest percentage of COD peak reduction occurred in the MSW reactors with MnP (M9) (92.28%), followed by M10 (88.89%), which produced the highest amount of methane. Figure 4-38 and Figure 4-39 show the percent reduction of BOD and COD of leachate of MSW reactors and food waste reactors after degradation respectively. In the food waste reactors, the greatest percentage of BOD peak reduction occurred in the reactor with cow manure (F4) (93.21%), but the highest percentage of COD peak reduction occurred in the F5 reactor with pig manure (71.2%).

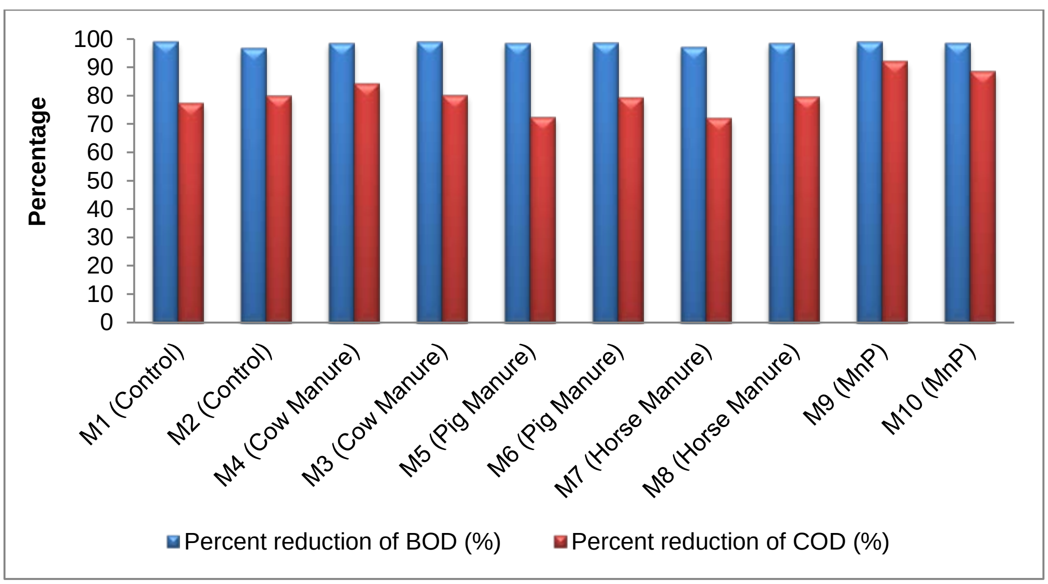


Figure 4-38 Percent reduction of BOD and COD of leachate of MSW reactors after degradation

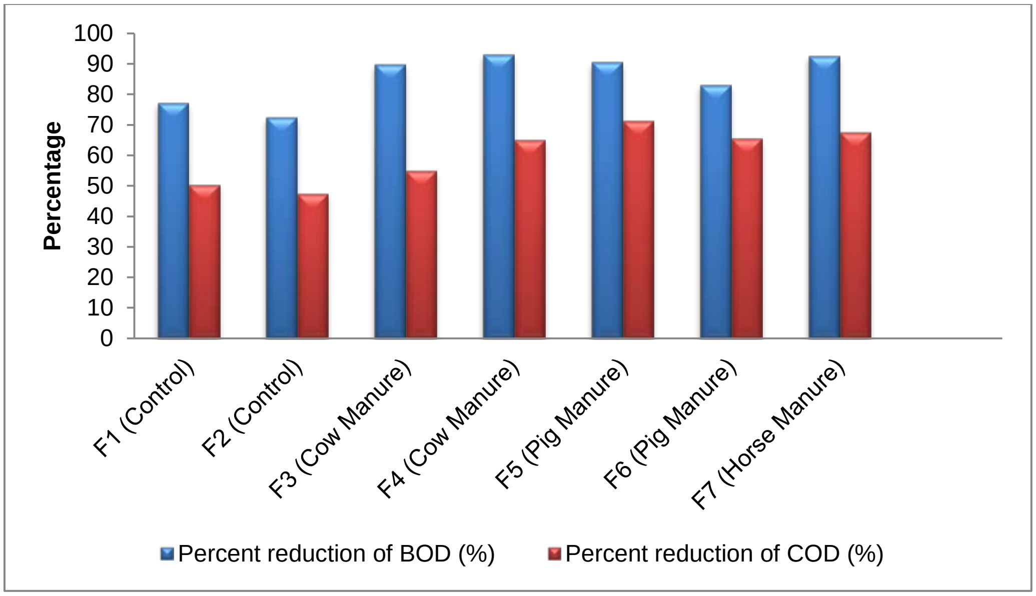


Figure 4-39 Percent reduction of BOD and COD of leachate of food waste reactors after degradation

4.7 Decay Rate (k) of Waste in Laboratory Scale Study

The rate of decomposition of organic material in the waste and the rate of landfill gas generation can be defined by the decay rate of waste. In the first order kinetic reaction, decay rate is defined as the biodegradation half-life of the organic material. Half-life ($t_{1/2}$) is the time to break down 50% of the original amount of organic material in the waste. According to Barlaz et al. (1990), the rate of degradation of solid waste in landfills depends on waste composition, waste particle size, moisture, ambient temperature, and pH. As the decay rate (k) value increases, the methane generation rate from landfills increases. In this study, the decay rate was calculated from the volume of methane generated from MSW and food waste reactors, as presented in Table 4-6. Details of the calculation procedure are described in Chapter Six. MSW reactors with MnP had the highest methane production as well as the highest decay rate of 4.3 and 5.8. The decay rate was significantly higher than the decay rates usually found in literature.

Table 4-6 Decay rate of waste in MSW reactors and food waste reactors

MSW Reactors	Decay Rate, k (Year^{-1})	Food Waste Reactors	Decay Rate, k (Year^{-1})
M1 (Control)	0.149	F1 (Control)	0.319
M2 (Control)	0.330	F2 (Control)	0.366
M4 (Cow Manure)	1.015	F3 (Cow Manure)	2.157
M3 (Cow Manure)	0.173	F4 (Cow Manure)	3.129
M5 (Pig Manure)	2.889	F5 (Pig Manure)	0.267
M6 (Pig Manure)	2.707	F6 (Pig Manure)	0.512
M7 (Horse Manure)	0.188	F7 (Horse Manure)	0.886
M8 (Horse Manure)	0.140	F8 (Horse Manure)	0.005
M9 (MnP)	4.340		
M10 (MnP)	5.859		

4.8 Summary

The results obtained from the laboratory scale landfill simulation in this study showed the feasibility of adding animal manure to waste feedstock to enhance biodegradation of waste and methane production. Various parameters of the waste degradation process, such as gas production, gas composition, leachate quality, and quantity were monitored, and the results aided in understanding the microbial behavior of waste and inoculum in the biodegradation process. It was concluded that pig manure and cow manure can be used as an effective inoculum for MSW and food waste, respectively. Though, the MSW reactors with MnP (M9 and M10) produced the highest amounts of methane, MnP may not be feasible for use in the field due to its very high cost.

Chapter 5

Construction and Instrumentation of Field Scale Biocell

5.1 Introduction

This chapter describes the procedures followed for construction of the field scale biocell and control cell at the City of Denton Landfill. It also includes the instrumentation of the monitoring equipment used to investigate the temperature and moisture of waste, leachate production, and landfill gas (LFG) generation. Construction of the two test cells in the field was a time-consuming process which required a number of tasks (with multiple steps associated with each task). The construction and instrumentation activities started on December 27, 2016 and were completed on January 23, 2017. The field scale construction of both the control cell and biocell began with waste sorting, to separate the organic waste as feedstock for the biocell. Various construction activities were executed on top of Cell Zero (Figure 5-1). The field activities involved excavation; cell placement in the excavated ground; filling the cell with waste; installation of leachate collection and recirculation systems, gas collection, and automated data collection systems; installation of moisture and temperature sensors; and setting up a solar panel. The tasks which were completed during the construction of the biocell are provided in Figure 5-2.

5.2 Study Area

The City of Denton Municipal Solid Waste Landfill is located on the southeast side of Denton, Texas, United States. It is the first landfill in the United States to conduct the biocell study. The location of the study area is presented in Figure 5-1. This location was selected because it is at the top of a closed landfill (Cell 0), is readily accessible, and the regular landfill activities would not be disturbed during the study period. According to the permit, the city of Denton has been receiving municipal solid waste since March, 1983. An area of 32 acres was separated for cell 1590, which is also designated as cell

0. Permit modification 1590A was completed in 1998. The Texas Commission on Environmental Quality (TCEQ) approved the process of recirculation of leachate and storm water to accelerate the gas production in 2009. Approximately 550 tons of municipal solid waste (MSW) is received by the landfill each year. About 80% of the received MSW is commercial waste, and 20% is residential waste. The landfill, which covers a land area of 252 acres, is divided into two sections: 152 acres for waste disposal and 100 acres for establishments. The landfill was installed with a leachate collection and recirculation system to operate as a bioreactor landfill.

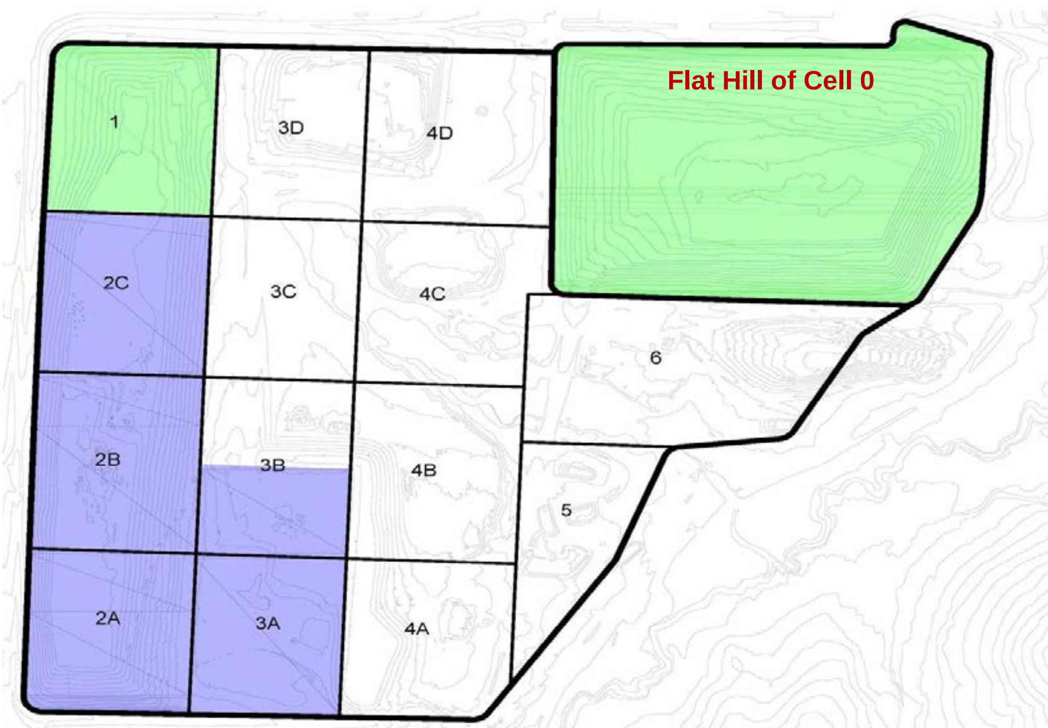


Figure 5-1 City of Denton Municipal Solid Waste Landfill and study location

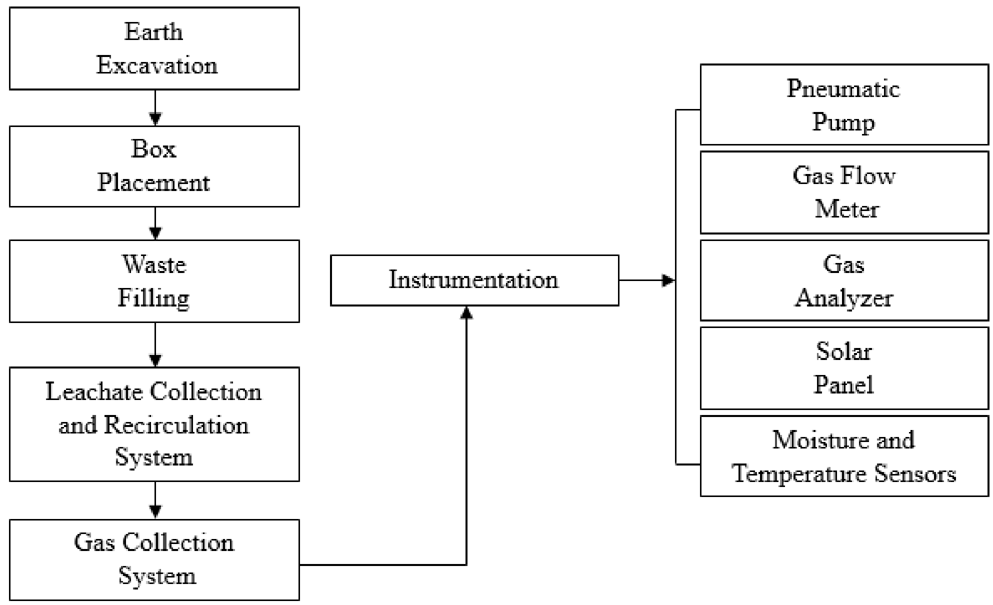


Figure 5-2 Sequence of biocell construction

5.2 Waste Sorting

From the residential waste that the city of Denton receives daily, organic waste was separated at the Building Material Recovery (BMR) facility, located in the City of Denton landfill. Residential waste was dumped on the BMR conveyer belt, where the inorganic plastic, glass, and Styrofoam were removed. Metal was sorted by the magnetic screening equipment at the end of the conveyer belt which separates the metal from the waste. The sorted waste was stored in three 50 cubic yard waste containers before filling the cells. Figure 5-3 shows the activities of waste sorting at the BMR facility at the City of Denton Landfill.

Figure 5-3 Waste Sorting at Building Material Recovery (BMR) Facility at City of Denton Landfill

5.3 Construction

5.3.1 Excavation

Soil was excavated at the flat hill of Cell-0 to install the boxes for the control cell and biocell, as shown in Figure 5-4 and Figure 5-5. The excavation area for each cell was made slightly higher than the actual box size (22 ft. x 8 ft. x 8 ft., or 6.71 m x 2.44 m x 2.44 m) for proper positioning of the boxes. Subgrades of the excavated areas were leveled to provide a smooth surface on which to place the boxes. Additionally, the subgrades of the excavated areas were slightly sloped (approximately 2%) so that the leachate can easily travel to the drainage port at the bottom of the box, accumulate in the leachate sump, and subsequently be pumped through the pneumatic pump. An additional area of 4 ft. x 4 ft. was also excavated, aligned with the longitudinal direction of the original excavated area to accommodate the leachate collection sump (Figure 5-7). The perimeter of the excavation area was marked with spray paint to ensure the required dimensions (22 ft. x 8 ft.) during excavation.

(a) (b)

Figure 5-4 (a) Measurement of excavated area; (b) Excavation of cell

Figure 5-5 Completed excavation area for cell placement

5.3.2 Cell Placement

Heavy equipment was required to place the cell underground. Figure 5-6 shows the placement of cell in the excavated ground. The cell was placed in such a way that it tilted slightly toward the drainage port. The placement of the cell was continuously monitored by survey equipment (a level), as depicted in Figure 5-8. After placement of the cell, soil was backfilled and compacted (Figure 5-9). Figure 5-10 shows the installed box after extensive earth work and backfilling.

Figure 5-6 Cell placement in the excavated area

Figure 5-7 Additional area excavated for placement of leachate collection sump

Figure 5-8 Sloping cell at drainage port

Figure 5-9 Backfilling around cell

Figure 5-10 Installed box after extensive earth work and backfilling

5.3.3 Waste Filling

Placing waste in a biocell is a sequential process that is similar to actual landfill cell filling. Waste filling in the cells was limited to residential solid waste to improve homogeneity of the material. Another benefit of using residential waste is that it typically has a high organic content, which improves gas production. The waste placed in the boxes was not compacted by earth-moving equipment. The wastes in the control cell and biocell were, therefore, allowed to compact by self-weight. Approximately 5 tons of residential MSW filled each of the cells, in three lifts of approximately 2 ft. each. Before placement of the MSW inside the box, the box was prepared to simulate the actual landfill. The preparation consisted of providing a drainage layer for leachate collection, setting up the gas collection system, and installing moisture and temperature sensors. The details of each of these activities are given in the following section.

5.3.3.1 Pea Gravel Placement

Drainage layers are constructed of materials which are highly permeable and sufficiently stable. Natural sand and gravel are used extensively in landfills for drainage purposes. Another significant criterion for drainage materials is that they be able to maintain a high level of hydraulic conductivity for long periods, to prevent plugging or clogging. Pea gravel was used as the drainage material for the bottom of the boxes in this study. Before it was placed, samples were collected from the source and were shipped to the laboratory so that the particle size distribution and hydraulic conductivity could be determined. Based on the grain size distribution curve, almost 85% of the materials were gravel, and the particle sizes were larger than 4.76 mm. Approximately 12% of coarse sand and 3% fine of sand was also observed from the distribution curve (Figure 5-11). No fine fraction was found in the samples collected from the source.

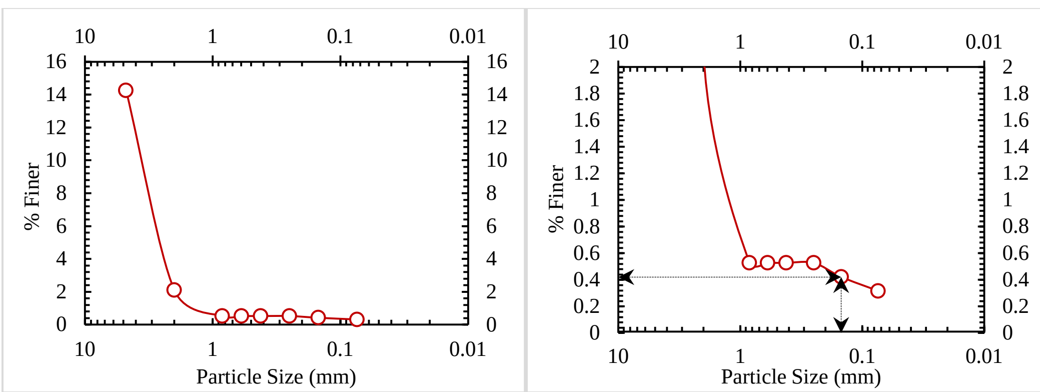


Figure 5-11 Grain size distribution curve for the pea gravel

From a wide range of laboratory test results, it is well recognized that the hydraulic conductivity of soils largely depends on the percentage of fines in it. Cedergren (1989) reported the influence of finer particles on the hydraulic conductivity of the materials (Table 5-1).

Table 5-1 Effect of fines on hydraulic conductivity of a washed filter aggregate

(Cedergren, 1989)

Percent Passing #100 Sieve	Hydraulic Conductivity	
	cm/sec	ft./day
0	3.0×10^{-2} to 1.1×10^{-1}	80 to 300
2	4.0×10^{-3} to 4.0×10^{-2}	10 to 100
4	7.0×10^{-4} to 2.0×10^{-2}	2 to 50
6	2.0×10^{-4} to 7.0×10^{-3}	0.5 to 20
7	7.0×10^{-5} to 1.0×10^{-3}	0.2 to 3

Based on the Cedergren (1989), if the percent passing through a No. 100 sieve is within a 2% range, the hydraulic conductivity ranges from 3.0×10^{-2} to 4.0×10^{-2} cm/sec. The measured permeability of the collected samples was 2.3×10^{-2} cm/sec. This value of permeability also satisfies the requirement for hydraulic conductivity for drainage layers specified by the US regulations. Based on the laboratory investigation of the pea gravel, it was selected as the drainage layer for the two boxes. The pea gravel was decanted into the boxes, using a bobcat (Figure 5-12), and a 10-in layer was placed on the floor of the cells. The gravel occupied a total volume of 143.33 ft^3 and was levelled with a rake (Figure 5-13).

Figure 5-12 Pouring pea-gravel in the cells

(a) (b)

Figure 5-13 (a) Flattening the surface of pea- ravel, (b) Final layer of pea gravel

5.3.3.2 Geotextile Placement

After placement of the 10 inch gravel layer, a 3.1 oz. nonwoven geotextile layer was placed over the gravel layer, as shown in Figure 5-14. This layer allows for filtration and drainage of leachate, and separates the waste layer from the gravel and drainage. The properties of the geotextile are showed in Table 5-2.

Table 5-2 Properties of 3.1 oz. nonwoven geotextile (erosionpollution.com)

Property	**Test Method**	**Roll Value**
Tensile Strength	ASTM-D-4632	80 lbs.
Elongation	ASTM-D-4632	50%
Trapezoidal Tear	ASTM-D-4533	30 lbs.
CBR Puncture Strength	ASTM-D-6241	175 lbs.
Mullen Burst	ASTM-D-3786	175 psi
AOS	ASTM-D-4751	#50 Sieve
Permittivity	ASTM-D-4491	2.2 Sec-1
Water Flow Rate	ASTM-D-4491	150 gpm/ft.
UV Resistance	ASTM-D-4355	70%/500 hours

Figure 5-14 Placement of geotextile drainage layer

5.3.3.3 Gas Collection Pipe Installation

Before placing the feedstock in the cells, three gas wells were installed in each cell, as shown in Figure 5-15 (a). The location of the gas wells are shown in Figure 3-15. The gas well is a 6 inch HDPE pipe, perforated on both sides. The gas collection system is described in detail in Section 5.3.6. A10-inch casing pipe was placed around the gas well, and gravel was used to fill in any bare spaces, as shown in Figure 5-15 (b) and (c). The main purpose of casing pipe was to hold the gravel around the gas well before the waste filling.

(a) (b) (c)

Figure 5-15 (a) Gas collection pipe installation; (b) Casing pipe placement around gas collection pipe; (c) Gravel placement inside casing pipe

5.3.3.4 Sensor Installation

A set of eight sensors were installed in the cell during the waste placement, to evaluate the moisture and temperature of the waste in the control cell and biocell. These sensors provided hourly readings that would were not collected during routine site visits. The data was stored in data logger. The sensor is described in Section 5.4.5. A sensor is usually installed in a landfill by boring the waste at specific depths, pushing the sensor into the waste and backfilling. For this study, however, the sensors were installed in a completely different way. Lids were placed on the cells, immediately after they were filled with waste, to facilitate reaching the anaerobic condition as quickly as possible. Boring of waste would have hindered the anaerobic condition, hence all eight sensors were installed before the cells were filled with waste. Because the sensors may move during the waste filling if they are not held securely, a frame of 1.5 inch PVC pipe was constructed to hold the sensors in specific locations in the cells during waste filling, as well as during waste decomposition. To construct the frame, a 1.5-inch PVC pipe was cut to various lengths and joined with a PVC connector, according to the length and height of the cells. It was designed so that the sensor cable remained in the pipe and the ports of the sensors were in contact with the waste. The frame was drilled in four locations to provide a way to remove the sensor ports from the pipe. After all of the sensors were attached to the frame, it was placed in the cell, as shown in Figure 5-17. Cables from the sensors were taken out from the lid of the cells through one of the 2-inch port on the lid and connected to the data loggers outside the cells.

Figure 5-16 Sensor attachment with PVC pipe

Figure 5-17 Placement of PVC pipe frame holding the sensors in the cells

5.3.3.5 MSW Filling

Once the pre-activities of waste filling were complete, the feedstock and inoculum were placed in the cells, as depicted in Figure 5-18 and Figure 5-19. The control cell was filled with sorted MSW, and the biocell was filled with sorted MSW and inoculum, according to the combinations shown in Table 3-2. Waste was not compacted by earth-moving equipment, but was compacted by self-weight. Each cell was filled with approximately 5 tons of residential sorted MSW, in three lifts of approximately 2 ft. each. During waste filling, water was added to the control cell to achieve a particular moisture content. Once the last layer of waste was filled, a 2-inch HDPE pipe for leachate recirculation was placed on top of waste, as shown in Figure 5-20.

Figure 5-18 Waste filling in the cell

Figure 5-19 Water addition during waste filling

Figure 5-20 Recirculation pipe placement after waste filling

5.3.3.6 Lid Placement

Once a cell was filled, a lid was placed on it immediately so that the cell would quickly reach an anaerobic phase (Figure 5-21). No additional solid (waste or inoculum) was added after the lid of the box was placed, which allowed for a first order rate of methane generation. The lids had three 6-inch ports to be used as gas wells and two 2-inch ports to be used as a leachate recirculation port and a channel for sensor wires. The design of the lid is shown in Figure 3-15.

Figure 5-21 Lid placement

5.3.4 Leachate Collection and Removal System

The function of a leachate collection and removal system is to collect the leachate generated within a cell and transport it to the leachate collection tank. It consists of a layer of drainage gravel across the entire base of the cell, a drainage geotextile layer on top of the gravel, and a 4-inch drainage port for a leachate collection sump. In this study, the leachate collection sump pipe was inserted into the drainage port of the cell prior to the cells being placed in the excavated ground. The drainage port was 4 inches above the base of the cell and 6 inches from the side that faces north. The leachate sump pipe was a 6-inch HDPE pipe that was welded to a 4-inch HDPE pipe (Figure

5-22). A 4-inch part was connected with the drainage port of the cell by a 4-inch threaded bolt, as depicted in Figure 5-23.

Figure 5-22 Leachate collection sump

Figure 5-23 Leachate collection sump connection with drainage port of the cell

A pneumatic pump was installed at the vertical sump pipes of both cells to extract leachate from the sump pipe. Details of the pneumatic pump are discussed in the instrumentation section of Section 5.4.1. Each of the cells had a leachate tank that was 44 inches x 38 inches x 38 inches. The leachate tank was placed on the top of a 5 ft. by 5 ft. by 2.5 ft. soil platform (Figure 3-15), which was located 3 ft. from the edge of the cell on the south side. The leachate tank was placed on the platform to maintain the gravity flow of leachate during recirculation. The base material was put in place and compacted by a bobcat compactor (Figure 5-24).

Figure 5-24 Construction of platform for leachate storage tank and placement

5.3.5 Leachate Recirculation System

The liquid recirculation system was operated in combination with the leachate collection and removal system to maintain optimum moisture content within the waste by recirculating leachate and any additional required moisture. The leachate recirculation system consisted of a 2-inch HDPE pipe that was perforated on one side at a 120 degree angle. The diameter of the perforation hole was 0.25 inches, and the distance between the two holes was 6 inches (Figure 5-25). The HDPE pipes were used to provide better flexibility under potential differential settlement. The horizontal perforated pipe was placed on the top of waste inside the cell, and the solid pipe connected the leachate injection port with the leachate tank placed outside the cell. The pipe network of the liquid recirculation system was designed to cover a maximum area from which to redistribute moisture within the waste mass. The main operational purpose of the liquid recirculation system is to maintain uniform moisture content throughout the biomass. When the control cell and biocell were operating, the average flow of liquid injection was kept approximately equal to the average flow of leachate in the leachate sump to prevent the biomass from becoming too wet or too dry. Figure 5-26 show the liquid injection pipe welding and perforation activities. Figure 5-27 shows the leachate collection and recirculation system of the control cell and biocell.

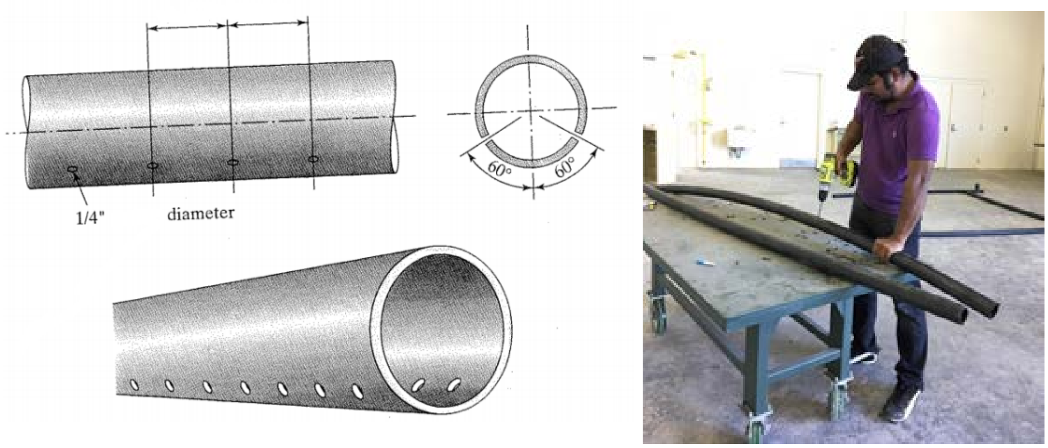


Figure 5-25 Leachate recirculation pipe perforation at 120 degrees

(a) (b)

Figure 5-26 (a) Leachate recirculation pipe welding and (b) Perforation testing

Figure 5-27 Leachate collection and recirculation system

5.3.6 Gas Collection System

The control cell and biocell were equipped with a landfill gas collection system for gas recovery. The lids of the cells prevented the release of any methane to the atmosphere and increased the efficiency of the gas collection system. Gas extraction was undertaken by vertical collector pipes and directed to the header pipe of the main gas collection system of the City of Denton Landfill, located southwest of the study site. Collected gas was conveyed under vacuum to the DTE facility through the wellheads and a network of header piping. The landfill gas (LFG) collection infrastructure consists of the following components:

- LFG extraction wells, including flow control valves, flow elements and monitoring ports.
- LFG transmission system, HDPE pipes, flow control valves, shutoff valve, gateway valve, flow meters, and monitoring ports.

5.3.6.1 LFG Extraction Wells

The vertical gas collection wells were 6-inch HDPE pipes which were perforated on both sides and installed during waste placement, as described in Section 5.3.3.3. The pipe was cut into the size mandated by the design of the cell, as shown in Figure 5-28. Vertical wells were wrapped by geotextile (Figure 5-29) and surrounded with coarse aggregate (pea gravel) to prevent clogging. Vertical wells protruded vertically through the 6-inch port on the lid, and wellheads (QED) were installed on the top of the ports (Figure 5-30a). The wellhead was an assembly of pipes and fittings that provided multiple functions, including flow adjustment, gas monitoring, and flow measurement.

Figure 5-28 Gas well cutting and perforation

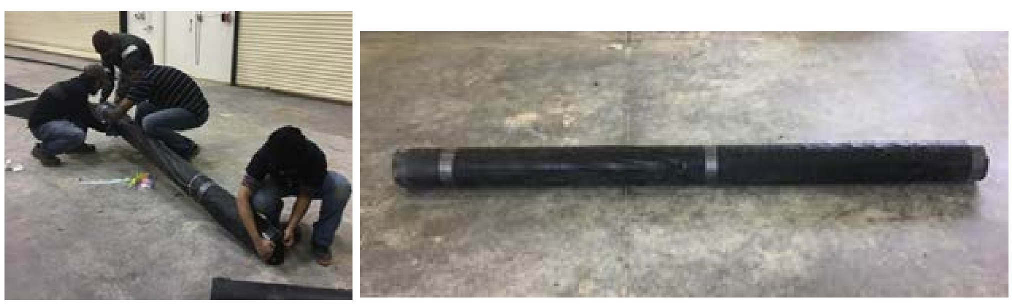


Figure 5-29 Geotextile wrapping around gas well

5.3.6.2 LFG Transmission System

The transmission system for LFG was mainly a set of HDPE pipes. Each wellhead was connected with a vertical 4-inch HDPE pipe that was connected to a 4-inch horizontal pipe that contained the flow meter for each box. This pipe was connected, with the main headers located on the southwest side of the study area with a shut-off valve.

The main function of the shut-off valve was to control the flow of LFG from the test cells to the main header line of City of Denton landfill. A thermal mass flow meter (FCI ST100) was used to measure total flow, flow rate, and temperature of the LFG extracted from the gas well. Figure 5-30b shows the transmission pipe installation, and Figure 5-31 shows the transmission pipe being joined with main header line of the City of Denton landfill .


(a) (b)

Figure 5-30 (a) Gas wellhead installation; (b) Transmission pipe installation


(a) (b)

Figure 5-31 (a) Shutoff valve connection at junction point of main header line; (b) Transmission pipe joining with main header line

5.4 Instrumentation

To evaluate the performance of the biocell, a pneumatic pump, air compressor, gas flow meter, solar panel, moisture and temperature sensor, and data logger station were installed. The details of the instrumentation are presented below.

5.4.1 Pneumatic Pump

A bottom-loader pneumatic pump (Viridian VP4-BL) was installed in the vertical sump pipe of each cell to extract the leachate accumulated in the sump pipe (Figure 5-32). A pneumatic pump is a pressure-operated pump that uses air pressure to extract the leachate/liquid from a sump well.

Figure 5-32 Pneumatic pump installation in sump pipe

The bottom loader is the standard pump type, which fills from the bottom and discharges through the outlet at the top. Its domed strainer arrangement allows the pump to easily bypass well ledges and obstructions during installation. A pneumatic pump requires an air compressor to inject air into the pump to extract the leachate accumulated in the sump pipe. A RIDGID 8-Gallon Gas-Powered Air Compressor was used in this study (Figure 5-33).

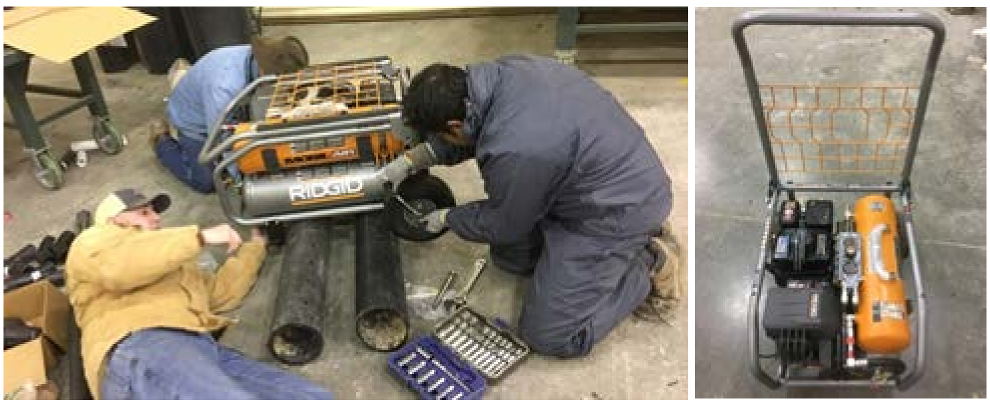


Figure 5-33 Air compressor set up

5.4.2 Gas Flow Meter

A thermal mass flow meter (FCI ST100) was used to measure the total LFG flow extracted from the control cell and biocell. This flow meter is the in-line type and measures the gas flow rate in standard cubic feet per minute (SCFM), totalized flow in standard cubic feet (SCF), and temperature in Fahrenheit (F). The flow meter was welded to the 4-inch gas transmission pipe. The main feature of this flow meter is that it can store up to 21 million readings. Data can be downloaded from the flow meter in MS-Excel format by ST100 Configurator Ver. 2.2.0.0 software. Figure 5-34 shows the installation of the flow meter.

Figure 5-34 Flow meter installation

5.4.3 Gas Analyzer

A Landtec GEM 2000 gas analyzer was used to monitor the composition of the LFG produced from both the cells by connecting it to the monitoring port of individual gas wellhead (Figure 5-35). This instrument measures the concentration of methane (CH_4), carbon dioxide (CO_2), oxygen (O_2), and balance gases.

Figure 5-35 Gas composition determination by Landtec GEM 2000

5.4.4 Solar Panel

An Ameresco Solar EFS 800 Series solar panel system was installed to supply power to the gas flow meters (Figure 5-36). This system consists of twelve rechargeable batteries that supply continuous 24 Volt DC (direct currents) to the flow meters. Three solar panels were set up to supply power to the batteries. The batteries can store power up to two weeks without being exposed to sunlight. Figure 5-37 shows the setup of the solar panel system.

Figure 5-36 Solar panel installation

Figure 5-37 Solar panel

5.4.5 Moisture and Temperature Sensor

Commercially available Model 5TM temperature/moisture sensors (manufactured by Decagon Devices, Inc.) were used to measure the temperature and volumetric water content of the municipal solid waste (MSW) in the control cell and biocell, as shown in

Figure 5-38. The sensor uses measurements of the dielectric constant to determine volumetric water content of the surrounding media and measures the volumetric water content (VMC) in m^3/m^3 once is in contact with the municipal solid waste. This sensor had been used in several studies to measure the moisture content of landfill cover soil (Alam, 2017; DeVries, 2016). The overall dimensions of the probes are 3.5 in. x 0.7 in. x 0.3 in. The 5TM sensors are capable of measuring moisture content that ranges from 0% to 100%, within error level of ±2%.

Figure 5-38 Moisture and temperature sensor 5TM (Decagon)

5.4.5.1 Sensor Calibration

Moisture content of solid waste is one of the most critical parameters for optimum performance of a biocell, as gas generation is largely dependent on the suitable moisture content of the MSW. Therefore, it was necessary to monitor the field measurements of moisture content in the control cell and the biocell. To ensure that the sensor will perform as required, it needs to be examined and calibrated before installation. Model 5TM soil temperature/moisture sensors can measure the volumetric moisture content of soil in m^3/m^3. This particular sensor had never been used for municipal solid waste (MSW), so custom calibration was required by laboratory testing to determine the field calibration factor. As MSW is a highly heterogeneous material, its composition affects the moisture content and unit weight. The same composition was used to represent the field condition as was used in the biocell (50% paper, 20% food, 15% yard waste, and 15% textile). After fresh MSW was collected from the working face of the City of Denton Landfill, it was sorted and separated (Figure 5-39), and the unit weight of the sample was measured by the standard compaction test (Figure 5-40). The compacted waste was put on a tray and pushed into the waste so that the sensor could measure the volumetric moisture content.

Figure 5-39 Sorted organic waste (a – Paper, b – Food, c – Textile, d – Yard)

Figure 5-40 Waste compaction

After the sensor measured the volumetric moisture content, the tray containing the sample was kept in an oven at 105 °C to measure the gravimetric moisture content

(GMC). There is a relationship between gravimetric moisture content and volumetric moisture content (Equation 5.1). Using Equation 5.1, the volumetric moisture content (m^3/m^3) was calculated from the gravimetric moisture content (dry unit basis) of the 24-hour oven-dried sample. The value was compared with the sensor-measured value and was plotted in a scattered graph to determine the field calibration factor.

Volumetric moisture content of solid waste,

$$\theta = \frac{Wd \times \Upsilon}{(1+Wd)\times \Upsilon w} \text{ .... (5.1)}$$

Where,

Wd = Dry gravimetric moisture content of solid waste

Υ = Unit weight of solid waste (pcf or kN/m^3)

Υw= Unit weight of water (62.4 pcf or 9.81 kN/m^3) (Qian et al., 2002)

5.4.5.2 Layout of Sensor

The Instrumentation was carried out in January, 2017. The layout of the sensors and details of the instrumentation are presented in Figure 5-41. Four sensors were installed in two rows and at two depths in each cell to monitor the water content and temperature of the waste. The sensors were connected to data loggers in the field to obtain continuous readings of the moisture content and temperature of the control cell and biocell.

5.4.5.3 Installation of the Sensors

Prior to the waste filling, four moisture sensors were installed at depths of 3 ft. and 6 ft. in each cell. Details of the installation of the sensors are described in Section 5.3.3.4.

5.4.5.4 Data Acquisition System

After the sensors were installed, the lead wires from the sensors were connected to an automatic data acquisition system to continuously monitor the moisture content and temperature. Two Em-50 data loggers were set up in the field to accommodate all of the sensors. The Em-50 is a 5-port, self-contained data logger that can measure the data in continuous intervals and can store up to 36,800 scans. The measurement interval for the current study was set to 60 minutes which allowed storing 24 data per day. The instrumented site, along with the data logger, is presented in Figure 5-42.

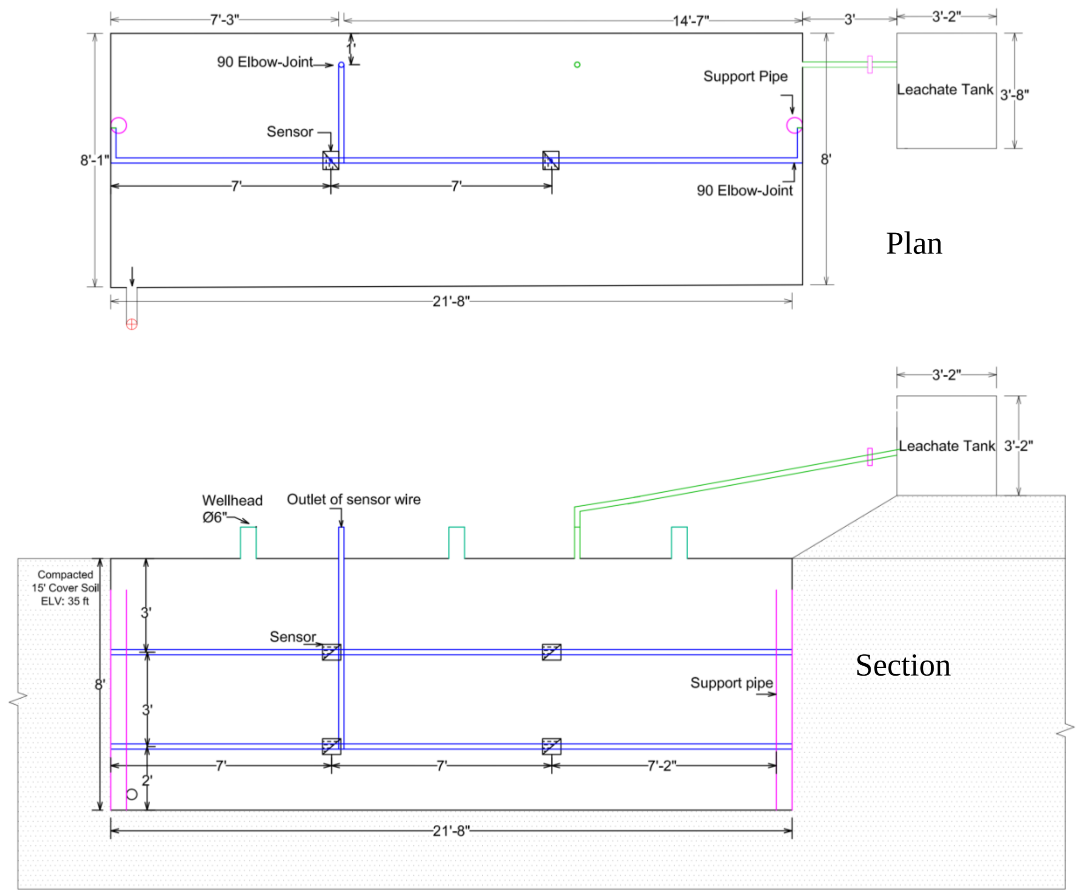


Figure 5-41 Sensor instrumentation layout

(a) (b)

Figure 5-42 Data collection (a) Location of data logger station, (b) Em-50 Data logger

5.4 Summary

The construction and instrumentation of a field-scale biocell that reflects an actual landfill in a smaller scale incur is challenging. To understand the sustainable waste management system, it was necessary to apply the biocell concept, in a pilot scale, in the field. Consequently, the construction and instrumentation of the field-scale biocell was one of the major components of this study.

Chapter 6

Performance Evaluation of Field Scale Biocell

6.1 Introduction

This chapter provides the results and analysis of field test cells: control cell and biocell from the field investigation and an automated data collection system in the field. The data was collected and monitored, beginning in January 2017, following field installation. The fundamental performance indicators of biocells are gas composition, flow rate and volume, leachate quality and quantity, and temperature and moisture content of the waste. Therefore, these characteristics have been monitored and analyzed on a regular basis in the field. Typical characteristics of waste that was placed in the biocell were determined in the laboratory at UTA. Performance monitoring was done to compare the biocell and control cell for gas generation, composition, leachate quality and quantity, temperature, and moisture content of waste.

6.2 Properties of Sorted MSW

The characteristics of the municipal solid waste used in the field test cells were determined in terms of physical composition, moisture content, and volatile solids, as shown in Figures 6-1 to 6-3. After sorting the organic MSW in the Denton BMR, as described in Section 5.2, six bags of samples were collected randomly from the organic MSW pile, and their physical composition, moisture content, and volatile solid content were determined by the procedure described in Section 3.4.5.1.

6.2.1 Physical Composition

Paper constituted the major portion of MSW in all the collected six bags. On average, the waste mass was comprised of 45% paper, 25% food waste, 16% textile, and 14% wood which was similar to the waste composition of the lab-scale reactors. Figure 6-1 shows the lab-determined physical composition of the sorted MSW.

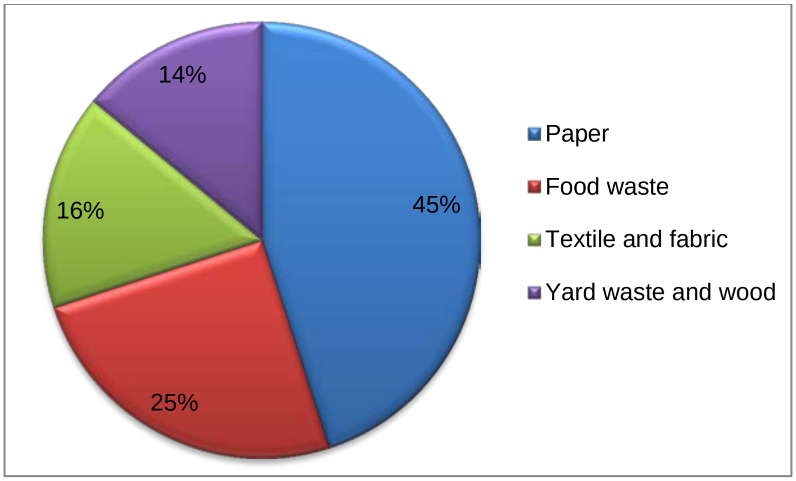


Figure 6-1 Physical composition of MSW

6.2.2 Moisture Content

Laboratory-determined moisture content of the sorted MSW was 35% to 54% on wet weight basis and 54% to 118% on dry weight basis, with high total solid (TS) content of 46% to 65%. The moisture content results on wet weight and dry weight bases are presented in Figure 6-2. The moisture content of fresh MSW depends on the composition of the waste, the season of the year, and the weather conditions. In this study, the waste mass used in the field test cells was collected in December, 2016 from Denton, Texas, USA. Among the six samples, the second bag had the highest moisture content (54.3%) on wet weight basis, as this bag had more paper and food waste than the other bags, which absorbed more liquid. The average moisture content on wet weight basis of the six fresh MSW samples was about 43%, which is pretty high compared to the average moisture content of fresh waste reported in other studies. Table 6.1 shows a comparison of moisture content found in this study with values reported in literature.

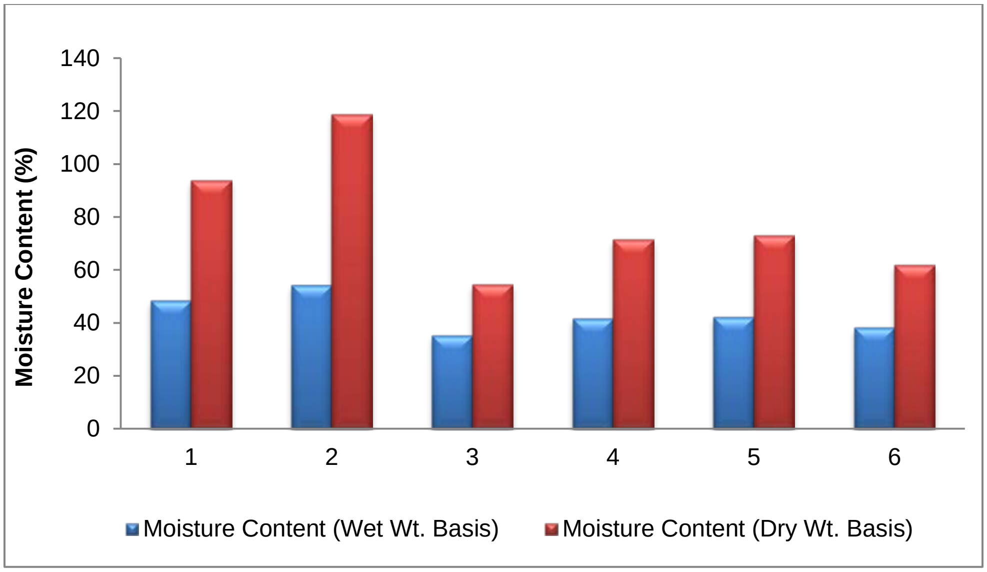


Figure 6-2 Moisture content (%) of MSW

Researchers suggest widely varying values of moisture content for fresh MSW. Pichtel (2005) suggested that the moisture content of MSW varies from 15 to 40% on wet basis, depending on composition, season of year, and the weather conditions in the United States. Reddy et al. (2009) estimated dry gravimetric moisture content of 44% for MSW collected from the working face of the Orchard Hills Landfill in the USA. According to Taufiq (2010), moisture content of fresh MSW from the working phase of the City of Denton Landfill in Denton, Texas, USA is 30% to 48% on wet weight basis. Shihada et al. (2013) estimated the moisture content of fresh MSW as 27.05%. In this study, the average MSW sample that was collected to be used in the laboratory scale reactors had a moisture content of 27.22%, which is pretty low compared to the MSW sample used in field test cells. This can be attributed to the season of collection. The MSW samples for reactors were collected in March 2016, in early spring. But the MSW samples used for field test cells were collected in December, 2016 (mid-winter).

Table 6.1 Comparison of moisture content of MSW in this study with literature

Reference	Moisture content (%) (wet weight Basis)	Remarks
Shihada et al. (2013)	27.05	Fresh un-compacted MSW
Taufiq (2010)	30 – 48	Fresh un-compacted MSW
Reddy et al. (2009)	44	Fresh un-compacted MSW
Pichtel (2005)	15 – 40	Fresh un-compacted MSW
Current Study (Lab-scale)	26.7	Unsorted MSW
Current Study (Field-scale)	35 – 54	Sorted MSW

6.2.3 Volatile Solid Content

The volatile solid content of sorted MSW samples was determined by burning the samples in a muffle furnace at a temperature of 550ºC for 2 hours, as discussed in Chapter 3. For the fresh sorted MSW samples, the volatile solid content varied from 75% to 83%, as presented in Figure 6-3. The average volatile solid content of the samples was 79.5%.

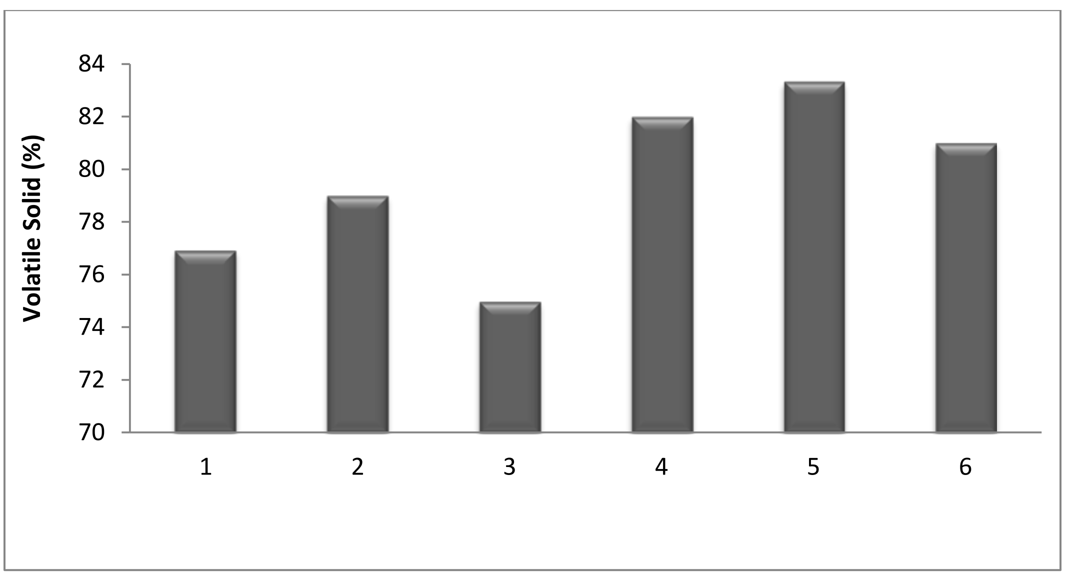


Figure 6-3 Volatile solid (%) of MSW

The results obtained in this study were found to be comparable to the results of previous studies. The result of the volatile solid content of this study is compared with those from literature and is shown in Table 2-11. Reddy et al. (2009) determined the volatile solid content of fresh MSW collected from the Orchard Hills Landfill in Illinois as 76% to 84%. Barlaz et al. (1990) estimated the organic content in fresh MSW as 79%.

6.3 Properties of Inoculum

Class B type sludge from a waste water treatment plant in Denton, Texas and pig manure from a local farm were used as inoculum for the biocell. Class B sludge is typically "undigested" and volatile; therefore, its use is not allowed in unrestricted public access areas or for cultivation of tobacco or leafy vegetables. Restrictive time limits are also established for certain activities after application. Duan et al. (2012) reported on mesophilic anaerobic digestion of sewage sludge of 23% TS, but the VS/TS ratio of their sludge was as low as 0.5–0.6. Guendouz et al. (2008) reported on mesophilic and thermophilic anaerobic digestion of high-solid municipal solid waste whose TS was 32%–40% and VS/TS ratio was around 0.4. Measured average moisture contents of sludge and manure in this study were 72.42% and 36.9% on wet weight basis, respectively.

6.4 Landfill Gas Characteristics

Landfill gas is one of the major indicators of waste decomposition. The field test cells began producing gas, mainly methane, carbon dioxide, and oxygen, soon after the completion of the field installation. To determine the composition and volume of gas produced from the test cells, a gas collection system was installed, as described in Chapter 5. This collection system was part of the landfill gas (LFG) management system of the City of Denton Landfill. It included vertical gas extraction wells, horizontal gas collectors (trenches), wellheads, valves, sub-header pipes, and a main gas header pipe that were connected with the main header line of the City of Denton Landfill. According to

Sperling (2009), a minimum amount of vacuum or suction at collection points (wellheads) is required to maintain an acceptable level of LFG collection efficiency, based on the system design, waste depth, well spacing, and the required radius of influence (ROI) of the LFG wells. Excessive levels of suction may result in air intrusion into the cells, which increases the risk of spontaneous combustion and landfill fire. An active gas extraction system, described in Section 3.5.1 and Section 5.3.6, was employed in both of the experimental cells. Figure 3-16 shows the locations of the gas extraction wells and their connection with the main header line. The system is maintained by a landfill gas-to-energy contractor, DTE, who collects and utilizes LFG-to-energy for the City of Denton Landfill.

A central suction controlled by the DTE Facility was applied from time to time to collect the landfill gas from the cells. The gas composition was monitored by GEM™ 2000+ on an almost-daily basis for the first two months. No suction was applied until an acceptable amount of methane was found in the composition. Once both cells started producing methane, 43 days after installation, suction was applied for the first time. After that, suction was applied twice a week to collect the gases, determine the gas composition and measure the volume. The recorded LFG volume and the volumetric percentage of methane were used to measure the total volume of methane. Monitoring frequency was reduced when the methane content and volume decreased.

6.4.1 Gas Composition of Field Test Cells

Landfill gas is generally composed of approximately 50% CH_4 and 50% CO_2 by volume, along with traces of other gases. Gas composition of the field test cells was monitored by a gas analyzer (GEM™ 2000+). The gas composition of the control cell and biocell are shown in Figure 6-4 and Figure 6-5, respectively. During the first month of monitoring period, in January 2017, the oxygen content was higher in both the control cell and the biocell, which indicated the existence of an aerobic condition in both cells. Methane content in both cells was very low in the first month. Carbon-dioxide content in both cells was not stable, and it was fluctuated frequently with the change in oxygen content. A sharp decrease in the carbon-dioxide content and an increase in the oxygen content were observed after 12 days due to an increase of oxygen concentration in both cells. This may have occurred due to a heavy rainfall and cold temperature which prevailed for a few days and could have resulted in settlement of soil around the cells and the intrusion of oxygen. There was another increase in oxygen concentration in the biocell after 18 days and in the control cell after 22 days. Concentration of carbon-dioxide and oxygen experienced a reverse trend throughout the one-month monitoring period of January 2017. After almost 40 days, the oxygen content in both of the cells started decreasing, which indicated the start-up of the anaerobic phase.

With the beginning of the anaerobic phase, the carbon dioxide content began increasing rapidly in both the control cell and biocell. In the middle of February 2017, the methane content in the biocell was just 8.47%. The biocell experienced more anaerobic activities than the control cell, based on the amount of carbon dioxide and methane. The biocell began producing methane earlier than the control cell, and the methane content reached almost 20% in the third month of monitoring (March, 2017). In control cell, methane content was always lower than biocell and it took almost five months to reach more than 20% methane for control cell. Oxygen content in biocell dropped to less than 1% in May 2017 and was stable for eight months after that. Oxygen content in control dropped to less than 1% in July 2017 in the control cell, showing that both of the cells reached the anaerobic phase.

The methane content in the control cell reached 45% in August, 2017. The highest methane content in the control cell was about 45.4% in August, 2017. Methane generation experienced two peaks in the last thirteen months of monitoring and began to drop from September, 2017. After that, methane content dropped very rapidly, falling to less than 10% in October, 2017. At the end of December, 2017, the oxygen content in the control cell began increasing with the decrease of both carbon dioxide and methane. This indicated that the decomposition of waste in the control cell was complete.

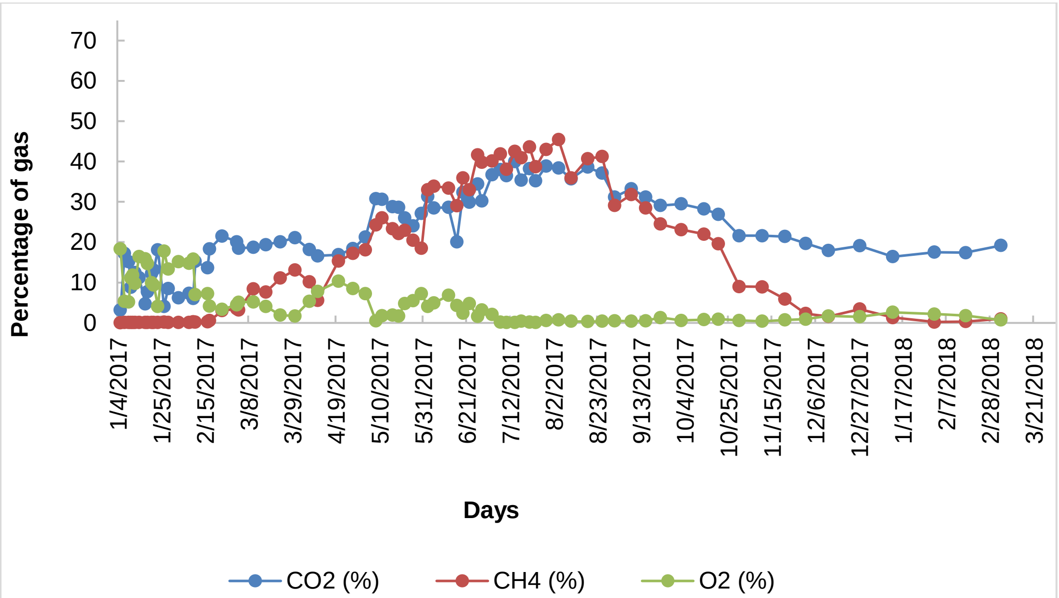


Figure 6-4 Gas composition of control cell

The methane content in the biocell reached 30% after four months and continued increasing. After almost 130 days of operation, it reached 40%; after 145 days (May 2017), it reached 45%; and in July 2017, it reached 50%. The methane content in the control cell never reached 50%. A study by Erses et al (2008) found that an anaerobic reactor didn't contain methane until day 165, due to the acidogenic conditions present. After that, methane began to appear and increased to 50% by composition. In the current study, the methanogenic phase began in the biocell after 145 days, when the methane content exceeded 45%. The oxygen content also supports that the biocell reached the methanogenic phase earlier than the control cell did. The methane content in the biocell peaked once during the last fourteen months of monitoring (63.3%) and began to drop very slightly at the end of October, 2017, showing that the methanogenic phase was longer in the biocell than in the control cell. At the end of December, 2017, the methane content in the biocell was 23.6%, and the carbon dioxide was decreasing as well. The lower values of carbon dioxide and methane in the biocell indicated that the waste was almost degraded.

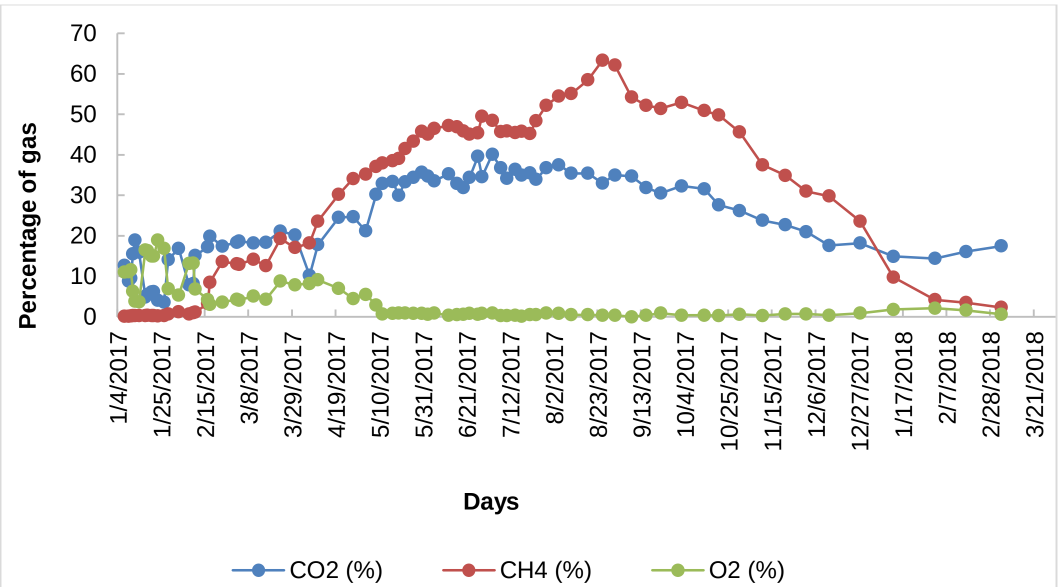


Figure 6-5 Gas composition of biocell

Figure 6-6 and Figure 6-7 show the CH_4:CO_2 ratio and the percentage of anaerobic activity (%) in the biocell and control cell, respectively. Both carbon dioxide and methane contents were low in the field test cells initially, so the ratio of CH4 to CO2 was almost zero until after 36 days in the biocell and after 44 days in the control cell. Both the carbon dioxide and methane content began increasing when the oxygen content started to decrease. The percentage of carbon dioxide started to decrease when the percentage of methane started increasing, which was after the acidogenic and transition phases were complete and the methanogenic phase began in the biocell and control cell. While the carbon dioxide content was as high as 60% in the lab -scale MSW reactors, it never reached more than 45% in the field. During the methanogenic phase, the ratios of CH_4 and CO_2 for all of the MSW reactors ranged from 1.3 to 2.8; while in the field, it rose as high as 1.9 in the biocell and 1.4 in the control cell. As in the lab scale study, 100% organic waste was used in the reactors, and the temperature was always controlled at 37^{o}C. The lab reactors experienced more methanogenic activities than the field test cells. The temperature of cells is greatly affected by the ambient temperature in the field, so it is difficult to achieve favorable conditions for methanogenic bacterial growth in field.

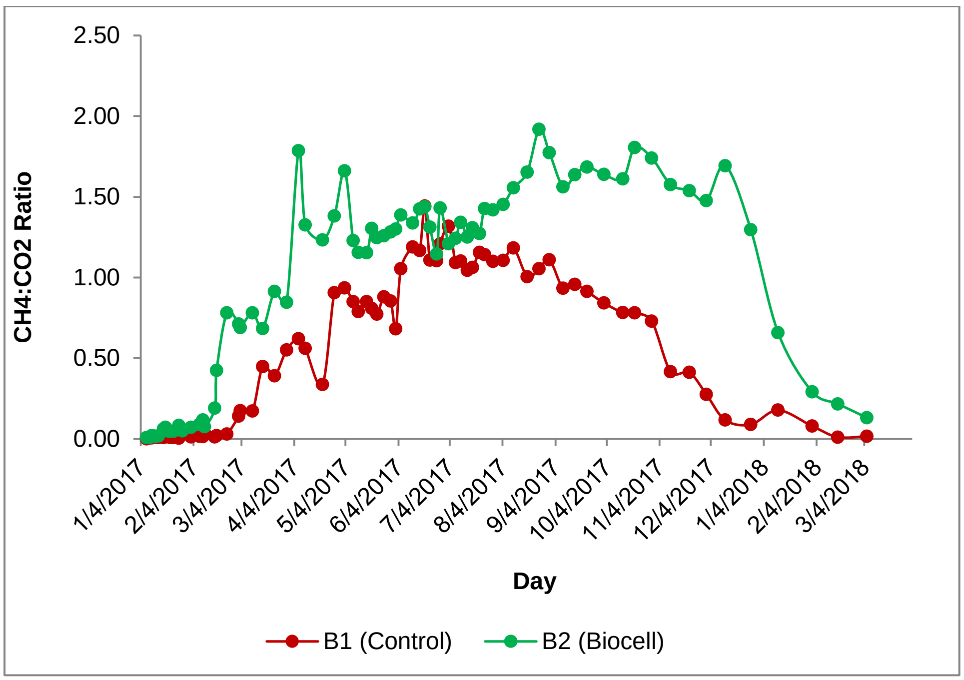


Figure 6-6 Methane to carbon dioxide ratio in control cell and biocell

The percentage of anaerobic activity in the biocell and control cell was calculated based on the Equation 4.1. The percentage of anaerobic activity in the biocell varied from 0% to 132%, whereas in the lab-scale MSW reactors, it was about 0% to148%, as described in Section 4.4.1. In the control cell, the percentage of anaerobic activity varied from 0% to 118%. Anaerobic activities increase with the increase of methane and decrease of carbon dioxide. The biocell achieved more than 100% anaerobic activities after just 3 months, but they started diminishing after approximately 12 months. The control cell achieved more than 100% anaerobic activities after 5 months, but they began diminishing after about 9 months.

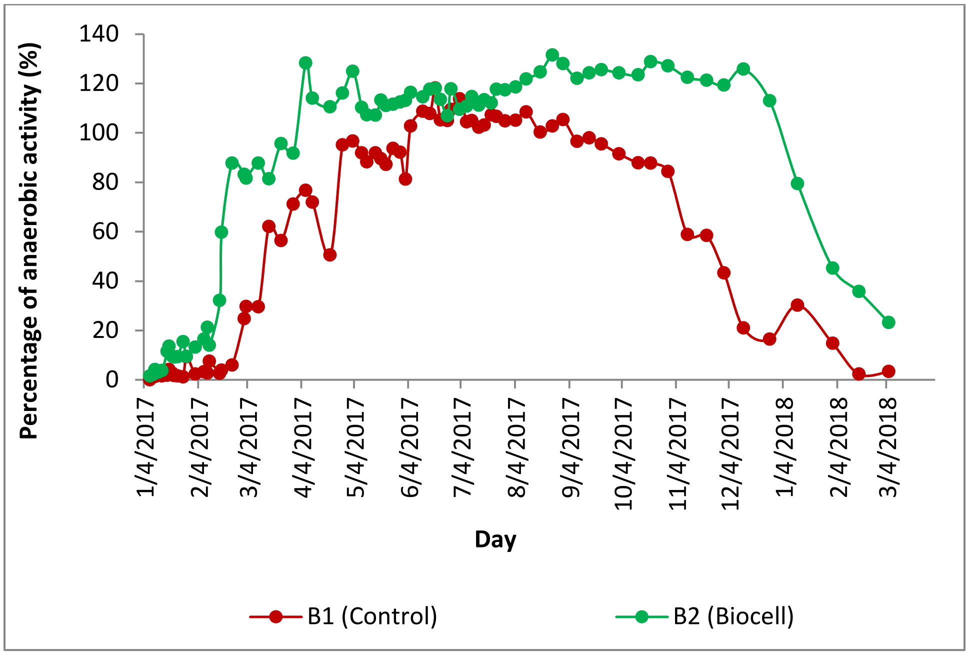


Figure 6-7 Percentage of anaerobic activity in control cell and biocell

In a study conducted by Erses et al. (2008), laboratory scale reactors were simulated at landfill condition and loaded with 19.5 kg of shredded and compacted solid waste mixture of 45% organic material (food + garden), 14.5% paper, 9.5% plastic, 5.6% textile, 3.8% glass, 2.2% metal, 4.4% ceramic, and 15% other materials (dust, wood, brick, miscellaneous) by weight. The study simulated the bioreactor landfill condition by recirculating leachate once per week, similar to this study, and operated for almost 640 days. Initially, the anaerobic reactor experienced some impedance in gas production, because of a long-lasting acidogenesis phase. The gas composition did not contain methane until the 165^{th} day. At around 330 days, the gas production began to increase, and the highest gas production was achieved on day 445 (20 L/day). During these days, methane content in biogas was about 48% (Figure 6-8).

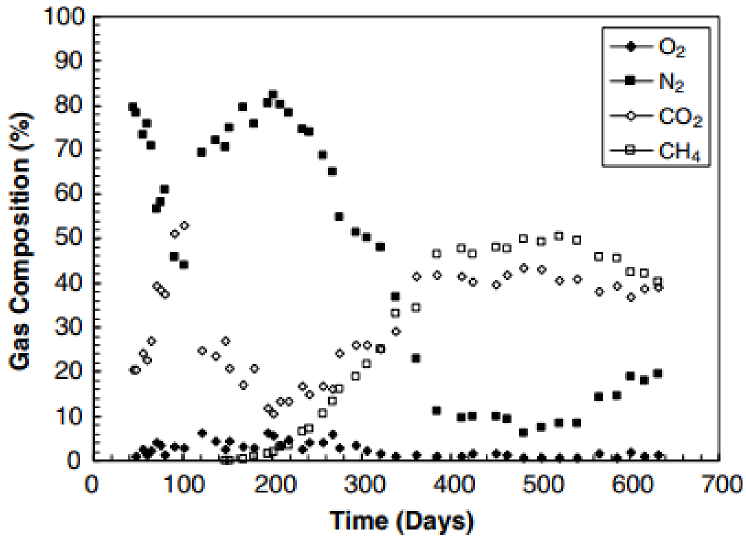


Figure 6-8 Gas composition in literature (Erses et al., 2008)

6.4.2 Gas Volume and Flow Rate of Field Test Cells

The landfill gas flow rate and volume are important indicators of the level of decomposition of the solid waste in the landfill. As mentioned in Chapter Five, the biocells and control cells used in this study were connected with the main header line of the gas collection system of the City of Denton Landfill, which is a central suction (vacuum) of landfill-gas-to-energy system. A flow control valve separates the main header line from the test section line.

Though the gas composition in biocell and control cell was measured immediately after installation and monitored, gas was not collected until a favorable amount of methane was found in the gas composition. The first suction was applied centrally after almost 43 days. Gas collection and extraction in the biocell and control cell began on February 17, 2017 but it was not fully operational until April, 2017 when the methane content in the biocell 30%. During the first month of monitoring, no suction was applied, as the methane content was very low in both cells. When the cells started producing methane, suction was applied in every two weeks, up to four months of monitoring. When the methane content in the biocell reached 30%, suction was applied twice a week. After 150 days of monitoring, about 3,088 standard cubic feet (SCF) of gas was produced in the biocell, with methane accounting for almost 1,165 SCF. Less gas overall (approximately 1,230 SCF) and a smaller amount of methane (270 SCF) were produced in the control cell after 150 days. After 424 days of monitoring, about 30,336 standard cubic feet (SCF) of gas was produced from the biocell, with methane accounting for almost 12,437 SCF of it. After 425 days, about 15,553 SCF of gas was logged in the gas flow meter from the control cell, with methane accounting for only 4,644 SCF of it. The cumulative volumes of gas and methane of the biocell and control cell are showed in Figure 6-9 and Figure 6-10, respectively.

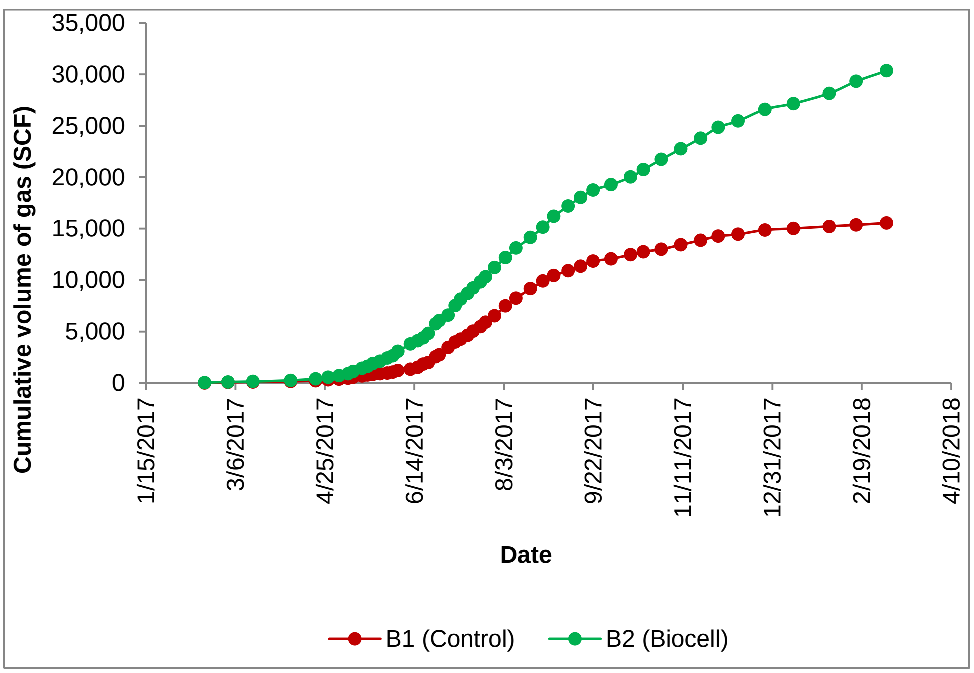


Figure 6-9 Cumulative volume of gas generated from control cell and biocell

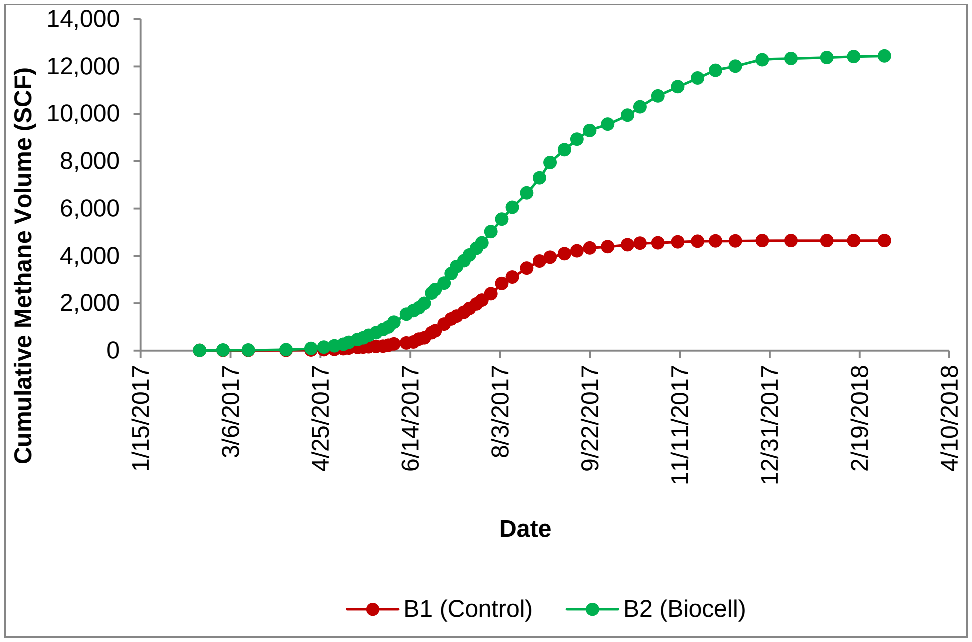


Figure 6-10 Cumulative volume of methane from control cell and biocell

The highest gas flow, 22 standard cubic feet per minute (SCFM), was seen in the biocell; the average was about 16.7 SCFM. In the control cell, the highest gas flow rate was about 21 SCFM, and the average was about 8.9 SCFM, which was almost half of the average gas flow rate of the biocell. Figure 6-11 shows the gas flow rates of the biocell and control cell.

Figure 6-12 shows the volume of methane generation throughout the monitoring period for the field-scale biocell and control cell. Methane generation is greatly affected by the weather conditions in Denton. The highest amount of methane was produced from the biocell and control cell during the hot summer of 2017, because the methanogens need a warm temperature to break down the waste. However, as the field test cells were monitored for only 15 months, the effect of temperature cannot be verified.

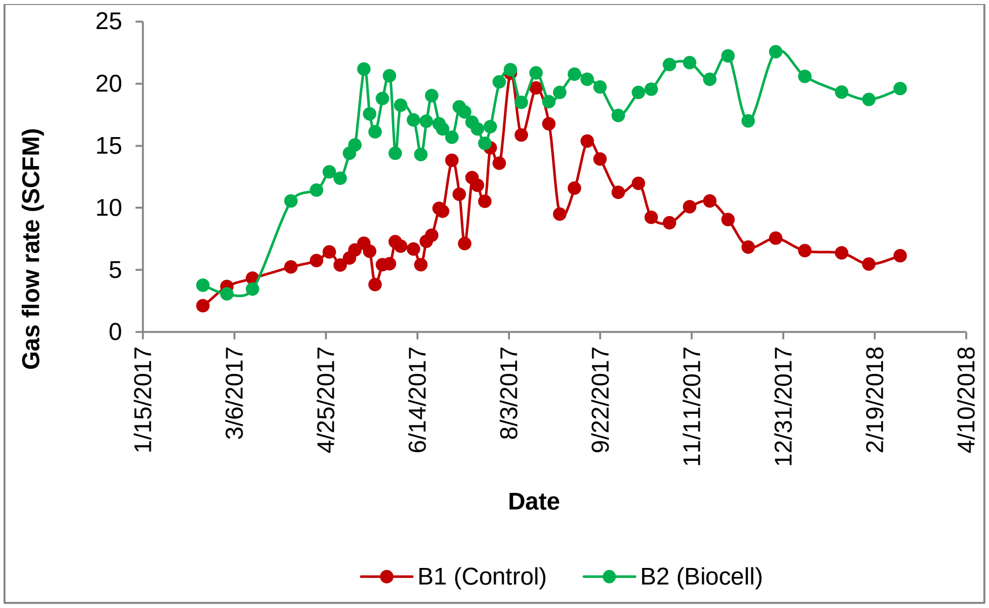


Figure 6-11 Flow rate (SCFM) of gas in control cell and biocell

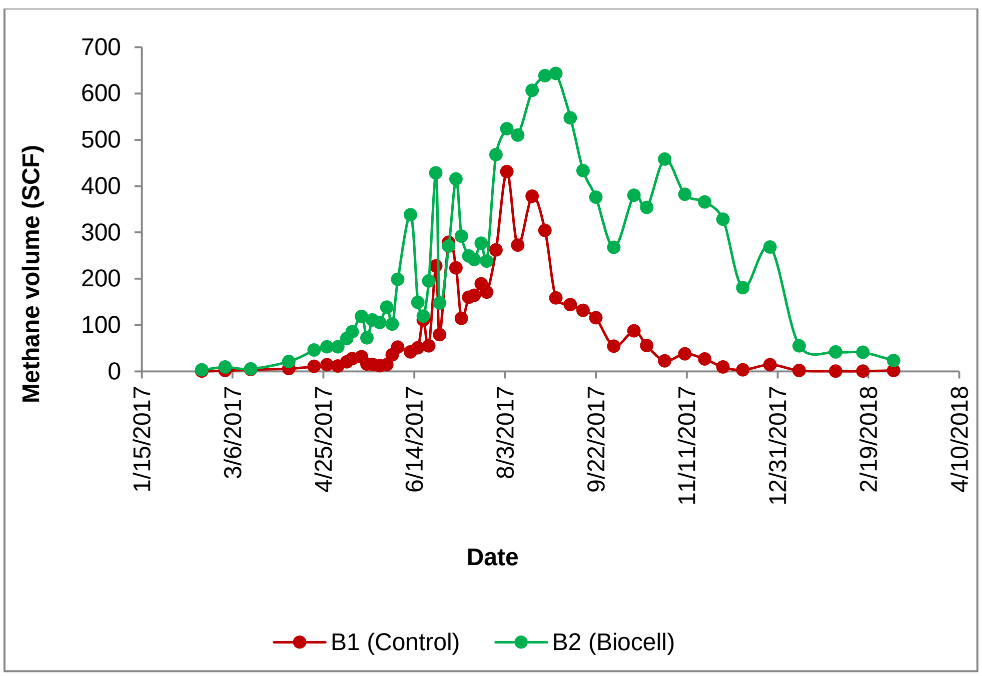


Figure 6-12 Volume of methane generation from control cell and biocell

Figure 6-13 shows the total methane yield of the field-scale biocell and control cell. The volume was converted to m^3/Wet Mg to compare with previous studies. About 117 m^3 of methane was produced from 1 Mg of wet waste in 424 days in the biocell, whereas about 29 m^3/Wet Mg was produced from 1 Mg of wet waste in 425 days in the control cell. A study was conducted by Yazdani (2010) at the Yolo County Central Landfill in California, where a field scale digester cell was operated in a two-stage batch system. The green waste was degraded under anaerobic conditions, followed by aerobic conditions, with the presence of aged horse manure as inoculum. During the anaerobic phase of 451 days, about 60,000 m^3 of methane was generated. The methane yield was 27.2 m^3 of methane per Mg of wet solid, as shown in Figure 6-14. From this comparison, it is comprehensible that the biocell produced almost four times the amount of methane than the control cell.

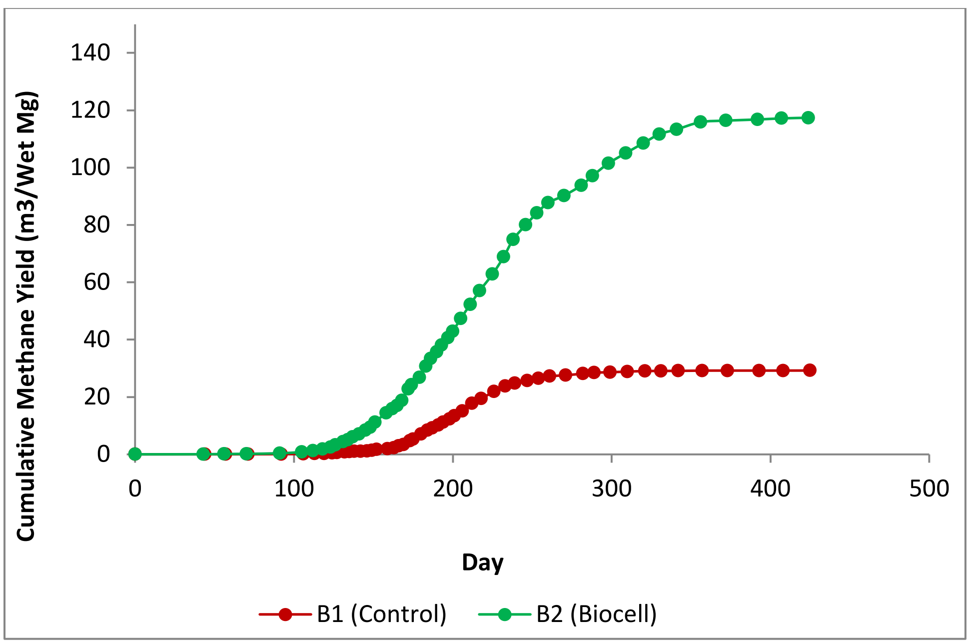


Figure 6-13 Cumulative methane yield in m^3/Mg from control cell and biocell

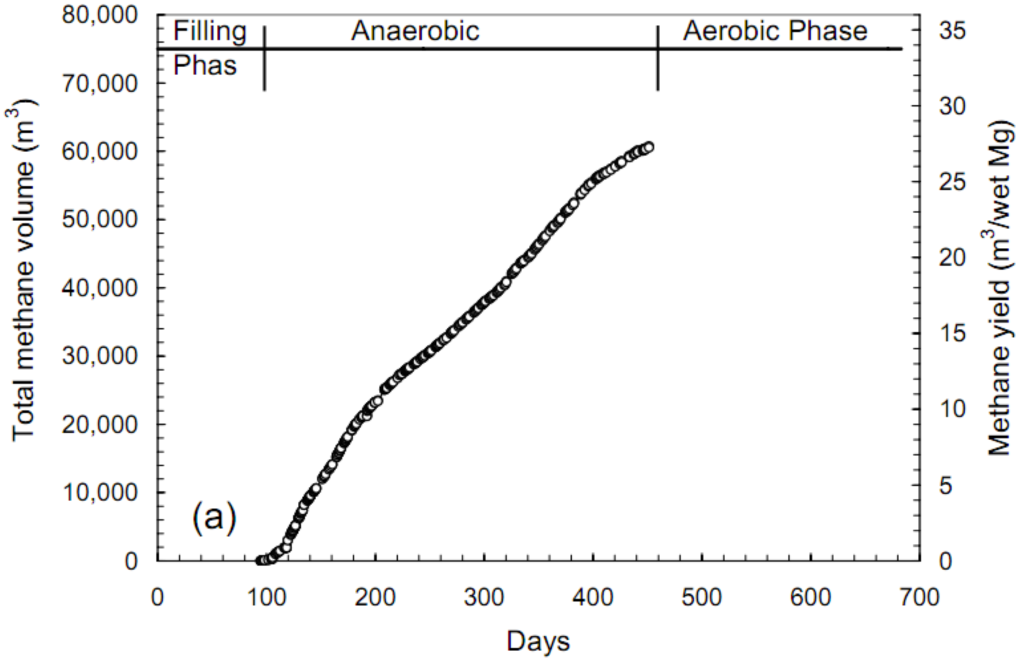


Figure 6-14 Total methane volume and methane yield for digester cell during anaerobic phase (Yazdani, 2010)

6.5 Leachate Characteristics

Leachate is another byproduct of the waste decomposition process. The chemical and biological processes of the waste are significantly influenced by the characteristics of the generated leachate, such as volume, pH, BOD (biochemical oxygen demand), and COD (chemical oxygen demand). The characteristics of generated leachate indicate the level of degradation of the solid waste. In this study, the volume of generated leachate, pH, BOD, and COD were monitored for both the control cell and biocell during their decomposition phases, as discussed in the following sections.

6.5.1 Volume of Leachate of Field Test Cells

During the first month of monitoring, the amount of leachate produced from the biocell and control cell was too low to be extracted by a pneumatic pump. A leachate sump port was placed 4 inch from the bottom, and approximately 10 inches of pea gravel (volume of 143.33 ft^3.) was placed on the bottom before being covered by a geotextile layer. The void ratio of gravel was 0.3 to 0.6 (Das, 2008), and it was filled with water to enable the leachate to reach the sump pipe. Assuming a void ratio of pea gravel of 0.45, 64.5 ft^3 or 482.49 gallons of water would be required to remain on the bottom. In the second month of the monitoring period, a pneumatic pump extracted a significant amount of leachate from both cells for the first time. After circulation of 150 gallons of water on day 7 and 300 gallons on day 28, 10 gallons of leachate were produce from the biocell and 18 gallons of leachate were produced from the control cell on day 36. The amount of leachate generation in the control cell surpassing that of the biocell supports the gas generation results. According to Alam (2016), when gas production is high, leachate production is less because the moisture inside the MSW is being used to produce gas. Throughout the monitoring period of thirteen months, gas production was higher and leachate production was lower in the biocell than in the control cell. The volume of leachate generation is shown in Figure 6-15.

Leachate generation is also governed by the amount of precipitation that falls on the landfill. In a study by Warith (2002), the leachate generation rate was approximately 25 - 30% of the total precipitation (Figure 6-16). But in this study, precipitation did not affect the leachate generation in either the control cell or the biocell, as both of the boxes were covered and sealed by a steel lid, preventing the intrusion of rainwater. The amount of leachate extracted was governed by the waste decomposition inside the cells, in-situ moisture content of the waste, and inoculum and addition of liquid.

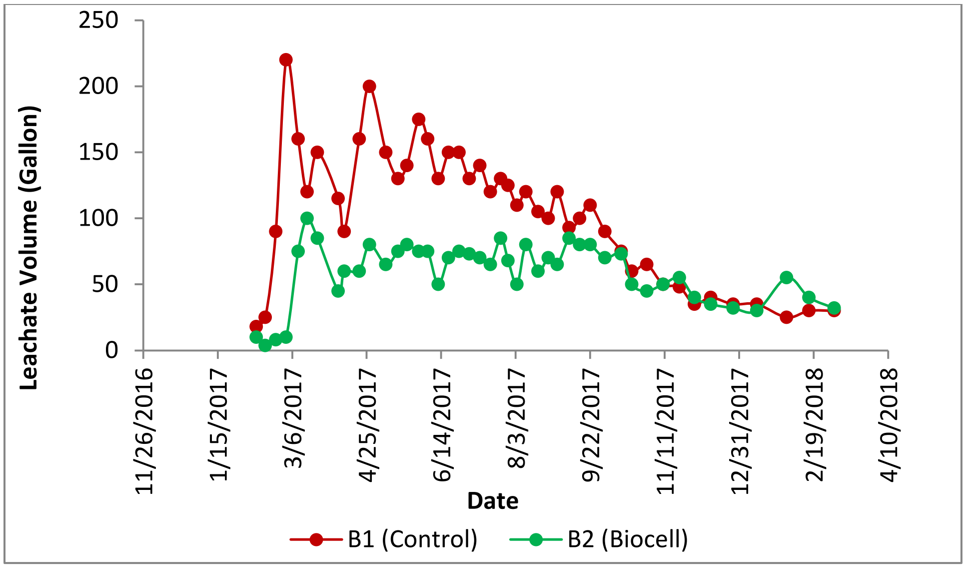


Figure 6-15 Volume of leachate generated from control cell and biocell

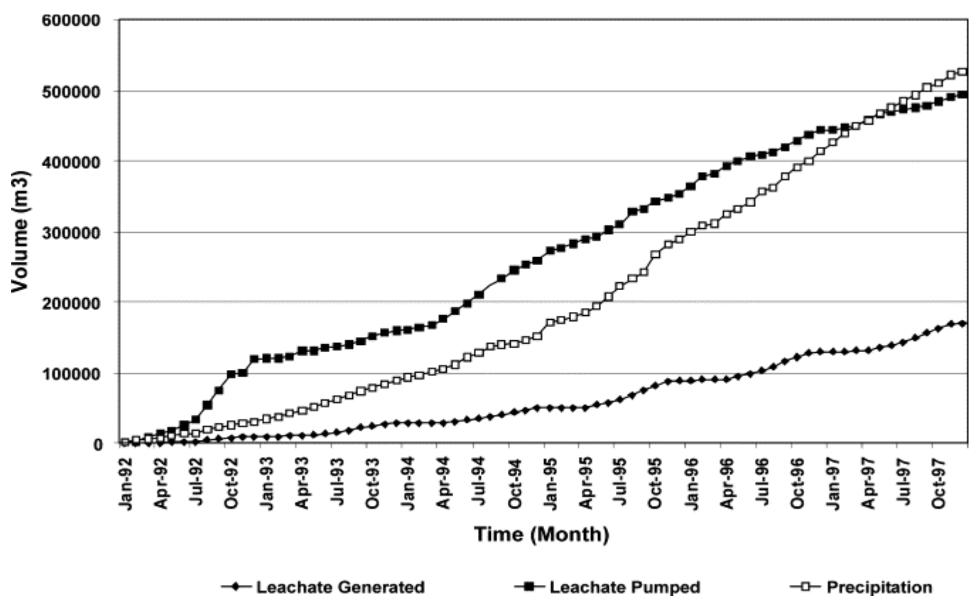


Figure 6-16 Leachate amount in Warith (2002)

6.5.2 pH of Leachate of Field Test Cells

The chemical and biological processes of the waste are significantly influenced by the pH of the leachate. The pH of the leachate of the control cell and biocell is depicted in Figure 6-17, which also shows the similar trends observed in the laboratory-scale experiments. In the first month of monitoring period, the pH of the leachate was more than 7, and that, combined with the presence of oxygen, showed that the cells were in the aerobic phase. The acidogenesis phase began in the second month, after the aerobic phase was completed and the pH had dropped below 7 due to the production of volume of volatile fatty acids. This was followed by the anaerobic decomposition phase of waste, and resulted in a decrease in pH from 8.02 to 6.06 in the control cell and from 7.94 to 6.73 in the biocell. As explained in Chapter Four, this indicated an increase in acetogenic microbial activity in both of the systems. The methanogenic phase started earlier in the biocell, which was indicated by a pH higher than 7. After four months, the pH of both of the cells was more than 7, indicating the methanogenic phase. The pH of both of the cells in the maturation phase was more around 8.

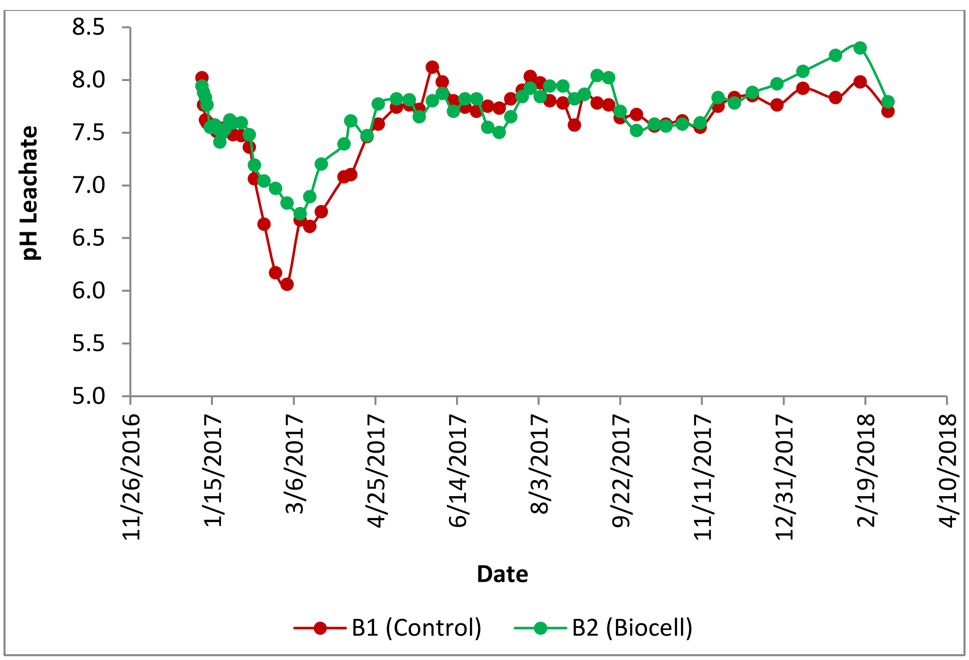


Figure 6-17 pH of leachate from control cell and biocell

Leachate pH is an indicator of different phases of MSW degradation. According to Reinhart and Al-Yousfi (1996), pH between 5.4 - 8.1 indicates Phase II (transition period); 5.7 - 7.4 shows Phase III, or acid formation. When the pH reaches 5.9, Phase IV, or methane fermentation, starts and continues until the pH is 8.6. A pH of 7.4 - 8.3 indicates Phase V, or final maturation. A study by Tatsi and Zouboulis (2002), found that the pH of leachate was more than 7.5 throughout the study period (Figure 6-18). In this study, the pH of the leachate in both cells exceeded 7 after a short acidogenic phase; then was around 7.5 during most of the methanogenic phase. This was similar to the results of the study by Tatsi and Zouboulis (2002).

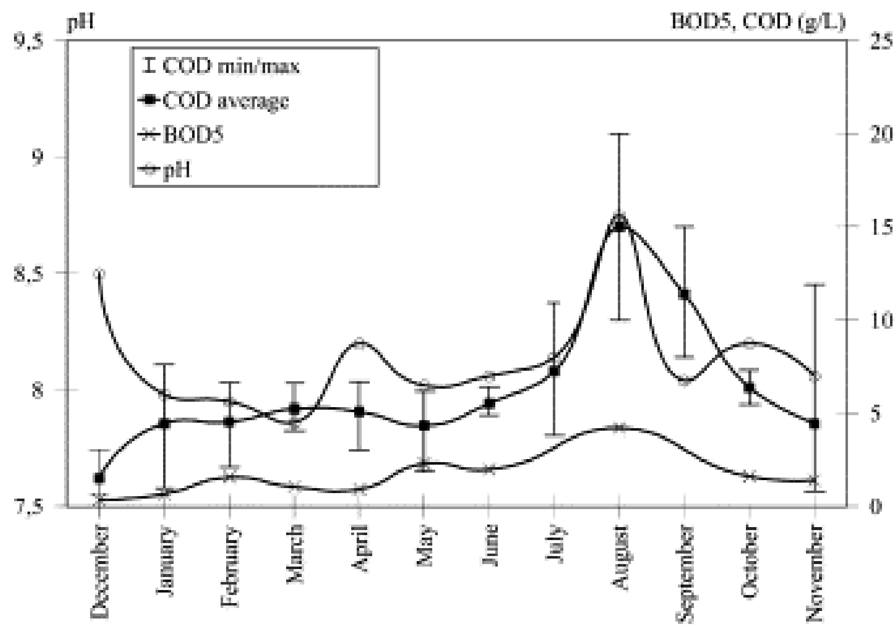


Figure 6-18 pH of Leachate (Tatsi and Zouboulis, 2002)

6.5.3 Chemical Oxygen Demand (COD) of Field Test Cells

The chemical oxygen demand (COD) of the leachate was measured on monthly basis to determine the level of degradation of waste inside the cells, as shown in Figure 6-19. Initially, the COD increased in both the control cell and biocell, indicating the beginning of the anaerobic phase, when there was not enough oxygen for the microbes in the leachate. The COD concentration increased until the methanogenic phase started, then dropped with the increase in the activity of methanogenic bacteria. According to Alkaabi et al. (2009), the COD value decreases initially due to the aerobic phase, and then increases. In the control cell, the COD value increased until the 4th month, then began to decrease; after 14 months, it was only 5,335 mg/L. Similarly, the COD in the biocell increased up to the 4th month; the highest value was 46,361 mg/L. After 14 months of monitoring, it was only 3,976 mg/L.

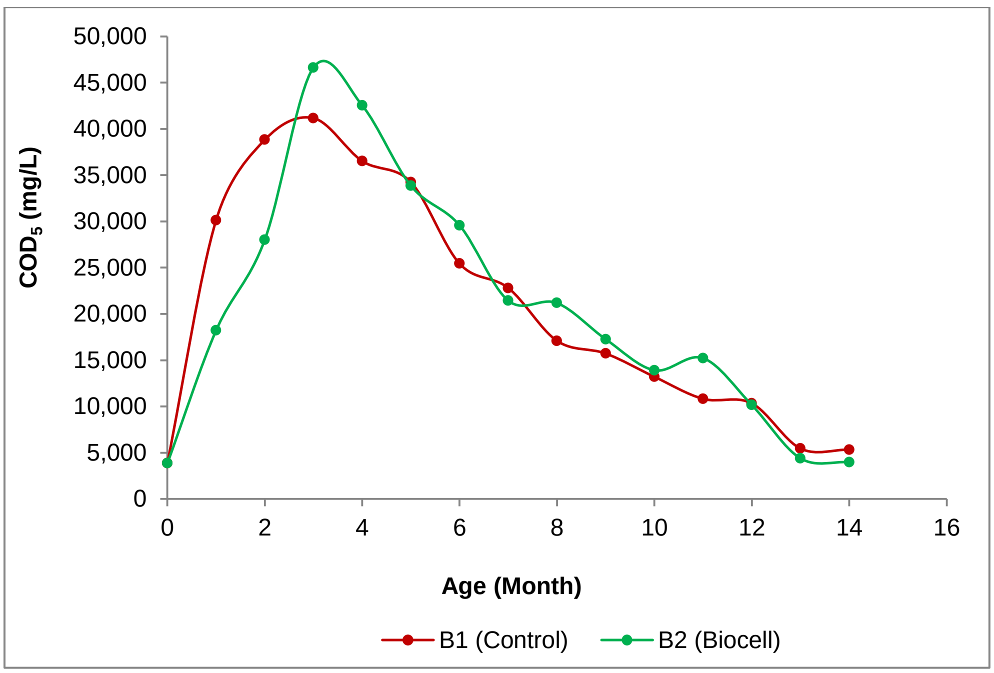


Figure 6-19 Chemical oxygen demand (COD) of control cell and biocell

6.5.4 Biochemical Oxygen Demand (BOD) of Field Test Cells

The BOD of the leachate in the field scale test cells was also measured monthly to determine the microbial concentration available to oxidize carbonaceous and nitrogenous compounds, as shown in Figure 6-20. The trend of the BOD value for both cells was similar to that of the COD value. A sharp increase was observed from the beginning until the 4th month for both cells, which showed that the methanogenic phase began in the 4th month. The initial BOD values of the control cell and biocell were 1,719 mg/L and 1,382 mg/L, respectively, and increased to 30,869 mg/L and 32,505 mg/L in the 4th month, then reduced to 291 mg/L and 217 mg/L, respectively, in the 14th month.

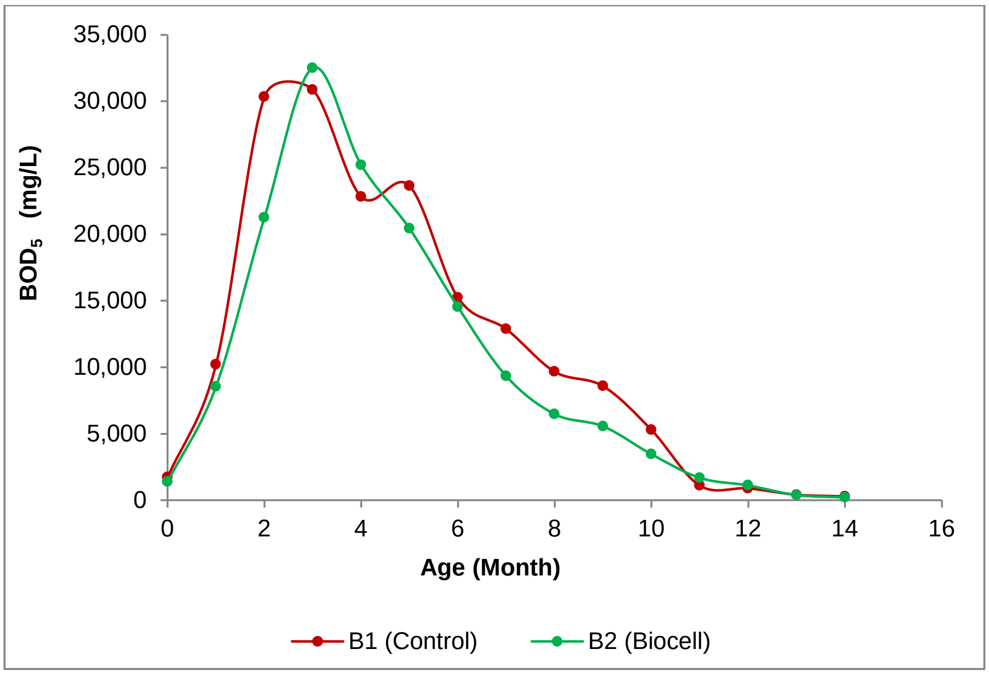


Figure 6-20 Biochemical oxygen demand (BOD) of control cell and biocell

Figure 6-21 shows the shows the BOD/COD of the leachate in the biocell and control cell. Initially, the BOD/COD ratios were about 0.36 to 0.45, which indicated the presence of biologically degradable organic matter in the feedstock. During the 3rd month of operation (March, 2017), the BOD/COD peaked at 0.76 in the biocell and 0.78 in the control cell. As the reactors started producing methane, this ratio was reduced, and at the end of the study, the ratio was the lowest (similar to the MSW reactors). The lowest values of BOD/COD (about 0.05) were measured at the end of the study, in the 14th month, indicating the lack of degradable material remaining in the leachate.

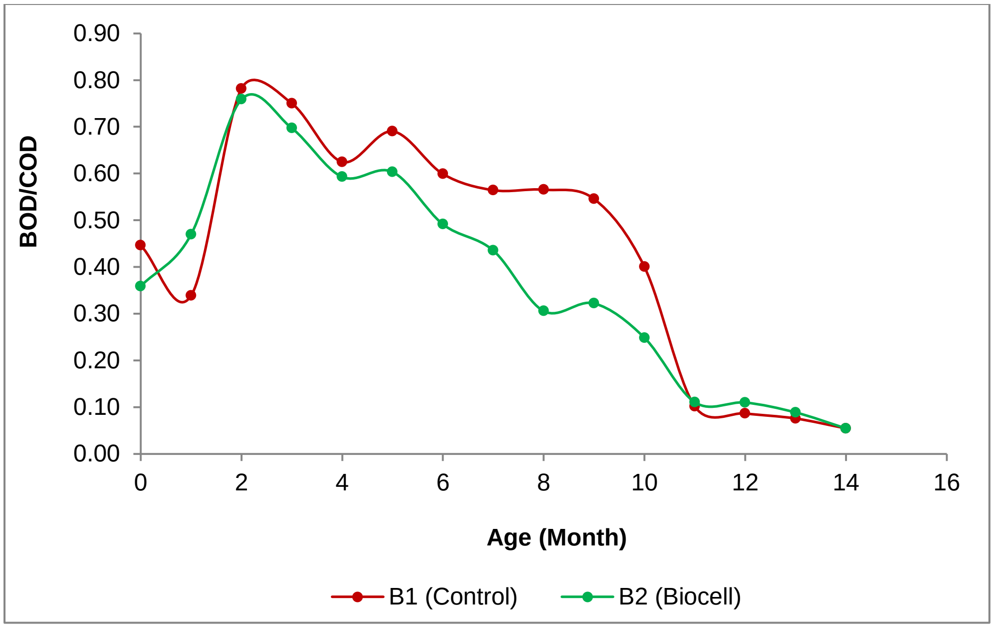


Figure 6-21 BOD/COD of leachate in control cell and biocell

The final COD and BOD of all of the leachate samples were measured at the end of the monitoring period. The percent of peak reduction of COD and BOD values were calculated based on the peak and end concentrations during the anaerobic process. In the control cell, the percentage of BOD peak reduction was about 99.06%, and in the biocell, the value was about 99.33%. The highest percentage of COD peak reduction (91.47%) occurred in the biocell, which produced the highest amount of methane. In the control cell, COD peak reduction was about 87.04%, leading to the conclusion that more decomposition occurred in the biocell.

6.5.5 Presence of Bacteria in the Leachate of Field Test Cells

The presence of methanogenic and other types of bacteria such as cellulolytics and acidogens, etc. in the leachate may provide an indication of the phases of decomposition of the MSW since their presence in fresh MSW (Barlaz et al., 1989) and is needed for methanogenesis. A protein test was conducted to identify the bacterial species present in the leachate of the control cell and biocell. The identified bacteria were divided into two categories and are presented in Table 6.2 and the appendix.

Table 6.2 Types of identified bacteria in the leachate of field test cells

Control Cell			Biocell		
Month	**Type of identified bacteria**		**Month**	**Type of identified bacteria**	
	Methanogenic	**Other**		**Methanogenic**	**Other**
January	1	4	**January**	3	3
March	0	9	**February**	1	1
April	1	2	**April**	2	3
June	1	3	**July**	3	13
October	3	3	**October**	3	7

Methanogens were found in the biocell throughout the monitoring period, indicating that more methanogenic activities were occurring in the biocell than in the control cell. The biocell also contained a greater variety of bacterial species. The volume of gas produced from both cells verified the results.

6.6 In-situ Moisture Content of Waste

The moisture content of waste inside the landfill is an important parameter for landfill operation. Bioreactor landfill technology relies on maintaining the optimal moisture content near field capacity (approximately 35% to 65%). It is able to do that by recirculating leachate or liquids, when it is necessary to maintain the ideal moisture percentage (US EPA). According to several studies, methane generation rate increases with an increase in moisture content (Warith et al., 2005), although it is extremely challenging to measure the in-situ moisture content of waste inside the landfill. Many in situ moisture-measuring devices have been developed for soil and tested for in situ moisture content measurement in landfills (Kumar et al., 2009). Six moisture-measuring sensors and techniques used for measuring the in situ moisture content of landfilled waste were reviewed by Imhoff et al. (2007): neutron probes, electrical resistivity (impedance) sensors, time domain reflectometry (TDR) sensors, partitioning gas tracer technique (PGTT), fiber optic sensors, and electrical resistivity tomography.

The initial moisture content of waste in landfills usually ranges between 15 and 35% by weight, and reported values of field capacity of MSW are 45 - 58% by weight (Remson and Fungaroli, 1968; Wigh, 1979; Walsh and Kinman, 1982; and Bengtsson et al., 1994). Some researchers suggest that sensor readings should not be used to measure the absolute moisture content because of the heterogeneous nature of MSW; rather, they can be used to measure changes in moisture content, with acceptable errors. If the in situ-density or unit weight of MSW is not known, errors may result in the conversion of volumetric moisture content from sensor readings to gravimetric moisture content in wet weight basis (Yuen et al., 2000). Continuous monitoring is therefore required to observe the trend in field moisture content measurement. In this study, moisture and temperature sensors were installed in the field test cells and were monitored continuously. The results from the sensors were analyzed and compared with the estimated moisture contents, based on the addition and removal of liquid from the cells.

6.6.1 Moisture Mass Balance

The mass balance of moisture in the cells was calculated based on the initial moisture content of waste, inoculum, and the addition of water. Precipitation was not considered. About 50 gallons of water were added to the control cell during the waste filling, and 25 gallons of water were added to the biocell. The sludge and manure in the biocell preserved some moisture, resulting in it needing less additional water. The initial moisture content in the biocell, including waste, sludge, and manure, was 45.7%; in the control cell, it was 43%. After box was sealed, water was added twice to both cells: 150 gallons on the 8th day and 300 gallons on the 29th day. After 150 gallon was recirculated, the moisture content in the control cell was 45% and in the biocell was 48% in wet weight basis. After the addition of 300 gallons, the moisture content in the control cell was 48% and in the biocell was 50% in wet weight basis. No liquid was added after that because plenty of leachate was produced. Figure 6-22 show the calculated gravimetric moisture content of waste in wet weight basis of the control cell and biocell.

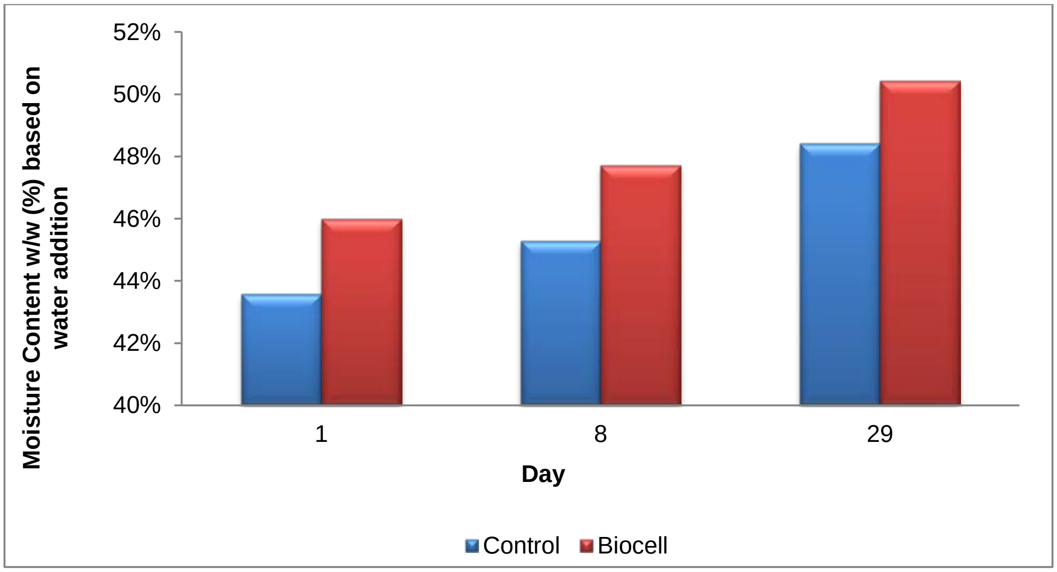


Figure 6-22 Moisture content of waste in control cell and biocell based on water addition

6.6.2 Sensor Results

For this study, four sets of in-situ moisture sensors were installed in each of the experimental cells. All of them were buried at depths of 3 ft. and 6 ft. Details of the instrumentation were described in Section 5.4.5.

The sensors were monitored throughout the study. Figure 6-23 shows the gravimetric moisture content in wet weight basis of the sensors in the control cell. Initial readings showed lower values of moisture content (33% to 42%), suggesting moisture content of waste similar to that tested in the laboratory. In general, the estimated gravimetric moisture content values, based on sensor readings, increased with the volume of liquid added during the first and second cycles of liquid addition (on days 8 and 29). About 150 gallons of liquid was circulated in the first cycle, and about 300 gallons in the second cycle. After the addition of 150 gallons on day 8th, a sharp increase in moisture content was observed from the sensor readings taken at 6 feet depth. Another increase was seen after 28 days, when 300 gallons of water was added.

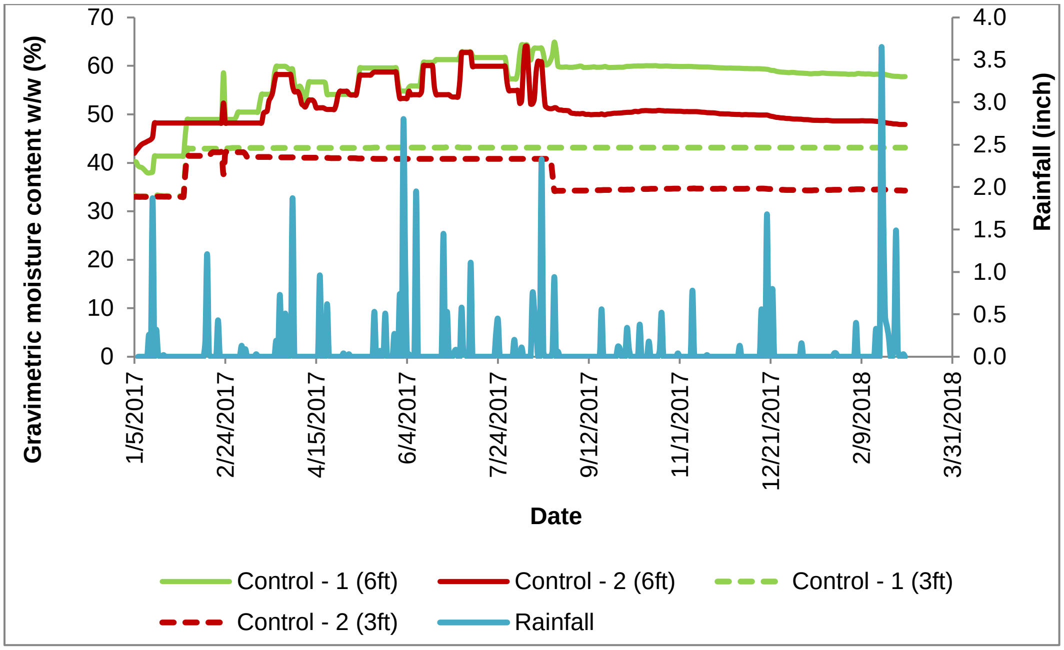


Figure 6-23 Gravimetric moisture content w/w in control cell

Daily average gravimetric moisture content in the control cell ranged from 32% to 64% in wet weight basis. Sensors at six foot depths showed changes in moisture content during the first eight months, but negligible changes were observed at three foot depths. From the beginning, collected leachate was recirculated to the cells once a week. After eight months of monitoring, all four sensors indicated negligible changes in the moisture content. Therefore, it can be inferred that the moisture content values achieved an approximate steady-state condition after eight months and did not change with continued recirculation.

Sensor data showed an initial increasing trend in moisture content in the biocell. As MSW is highly heterogeneous, the data varied significantly for the four locations, but the trends were the same. Figure 6-24 shows the gravimetric moisture content in wet weight basis in the biocell. At a depth of six feet, the moisture content was highest in the tilted area that was close to the drainage port. Liquid was added in two cycles, on days 8 and 28. The moisture content at a depth of 3 feet sharply increased after the addition of 150 gallons of water. After the addition of 300 gallons of water, there was another sharp increase in moisture content, at depths of both 3 and 6 feet. Based on the sensor readings, the daily average gravimetric moisture content in the biocell ranged from 34% to 63% in wet weight basis. There is no effect of rainfall on moisture content seen in both the control cell and biocell. Both of the cells were covered and sealed by a steel lid, so there is no chance of rainwater intrusion in the cells.

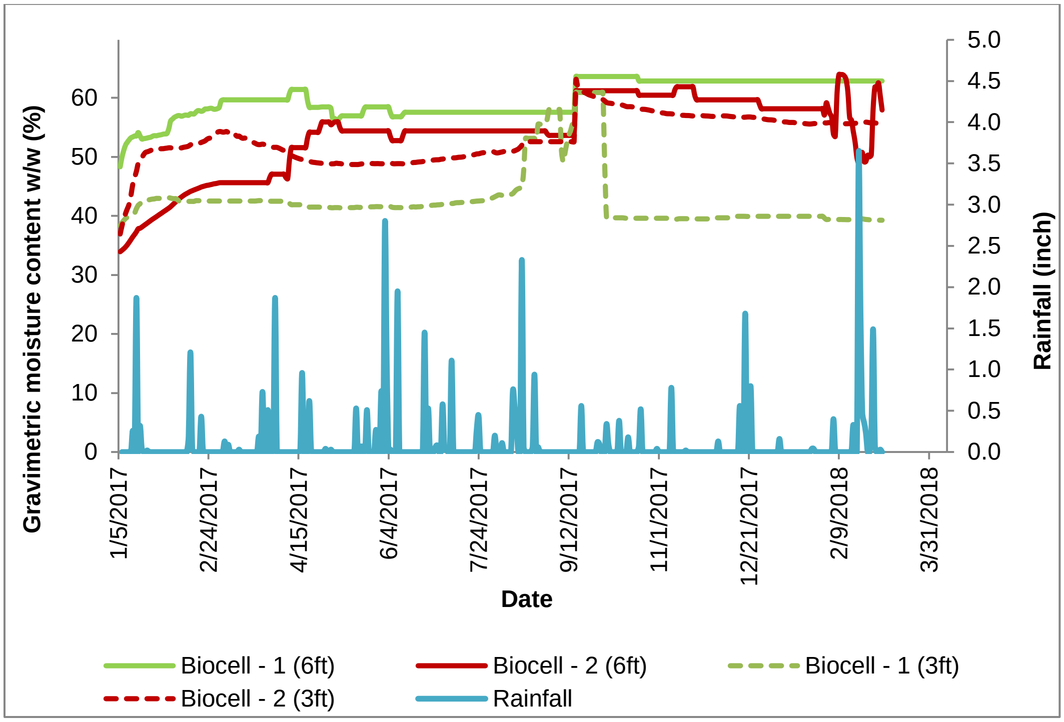


Figure 6-24 Gravimetric moisture content w/w in biocell

In a study conducted by Yazdani (2010), the average moisture content of a pilot scale digester cell was 59.2 + 0.2% during the anaerobic phase of operation, which is comparable to the moisture content of this study. The moisture content increased when the leachate was recirculated, but at a certain point, both of the cells reached a steady-state condition, where frequent recirculation did not affect the change in moisture content. A similar pattern was also observed in a study conducted by Kumar et al. (2009) where, the moisture content values, predicted based on sensors, attained an approximate steady-state condition after around 750 days of monitoring and did not increase with continued liquid addition. In that study, the moisture content over the duration of the study was estimated based on laboratory-derived and field-derived calibration curves. Based on information from the moisture sensors, the spatial average moisture content values were estimated, and it was observed that they increased gradually during the period when liquid was added (days 150–870) in both the laboratory-calibration and field-calibration curves as shown in Figure 6-25 (Kumar et al., 2009).

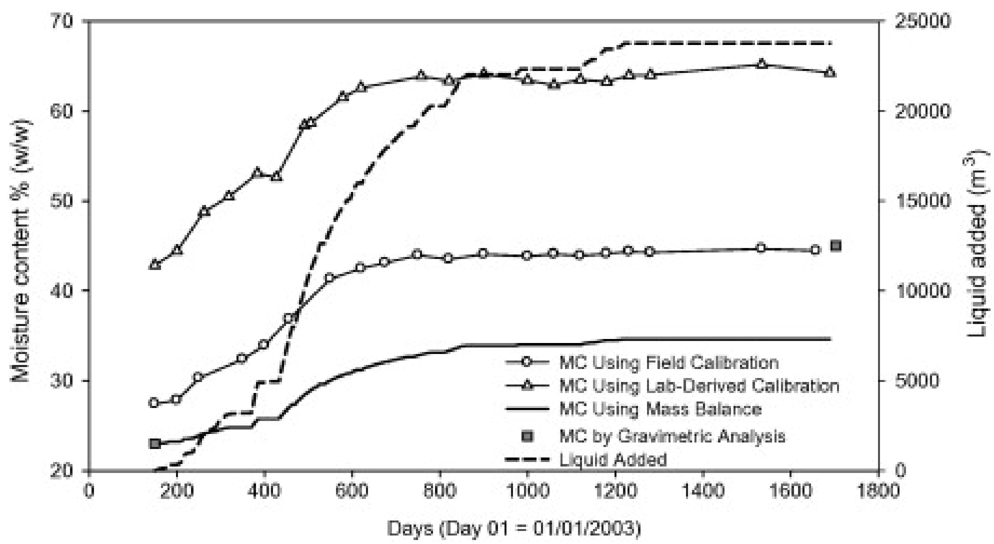


Figure 6-25 Comparison of temporal variations of moisture content estimated using sensors and mass balance approach (Kumar et al., 2009)

6.7 Temperature of Cells

6.7.1 Ambient Temperature

Denton, Texas has a humid, subtropical climate with hot, humid summers and cool winters. It lies on the southern end of a tornado zone, commonly referred to as "Tornado Alley," and tornado watches are issued occasionally by the National Weather Service, although tornadoes usually form outside the city. According to US Climate Data, ambient temperatures from January, 2017 to March, 2018 ranged from a low of 2.9 °C to a high of 29.2 °C. The average ambient temperature of Denton during the monitoring period from January, 2017 to March, 2018 is shown in Figure 6-26.

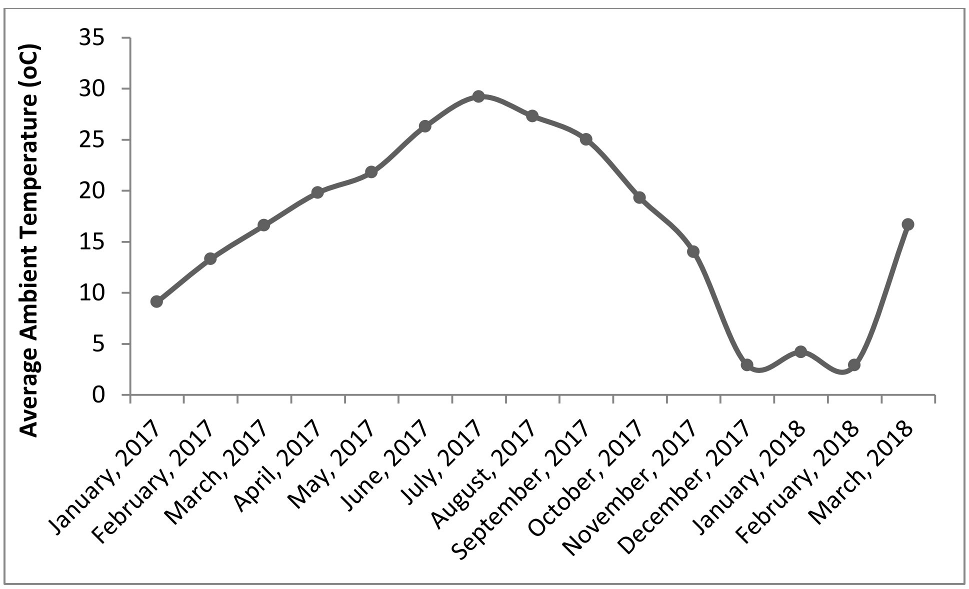


Figure 6-26 Average ambient temperature in Denton, Texas during monitoring period (usclimatedata.com)

6.7.2 Temperature of Waste in the Cells

Temperature is an important factor that determines the rate of digestion. According to various studies, temperature can be an important determinant of biological activity. The in-situ temperature of waste in a landfill is mainly governed by the balance between the rate of heat production due to biological activities and the rate of heat loss to the surrounding environment (e.g., soil, groundwater, and atmosphere). While less heat is produced in the anaerobic process than with aerobic metabolism, Rees (1980) demonstrated that if a cell is well insulated against heat loss, a well-established methanogenic system can sustain a much higher waste temperature than the ambient, even in a temperate climate. In this study, eight sets of in-situ temperature sensors were installed in the experimental cells. The detail of the instrumentation of the sensors was described in Section 5.4.5. Temperature data obtained from the sensors was plotted for each day, and the average daily temperature for each layer (at two depths) of the control cell and biocell are shown in Figures 6-27 and 6-28, respectively. The temperature readings showed similar trends of both cells at the same depth. An increasing temperature trend was seen in the first of monitoring due to biological activity in the waste. A decrease in temperature was seen at six feet depths in both cells, four days after the first water addition.

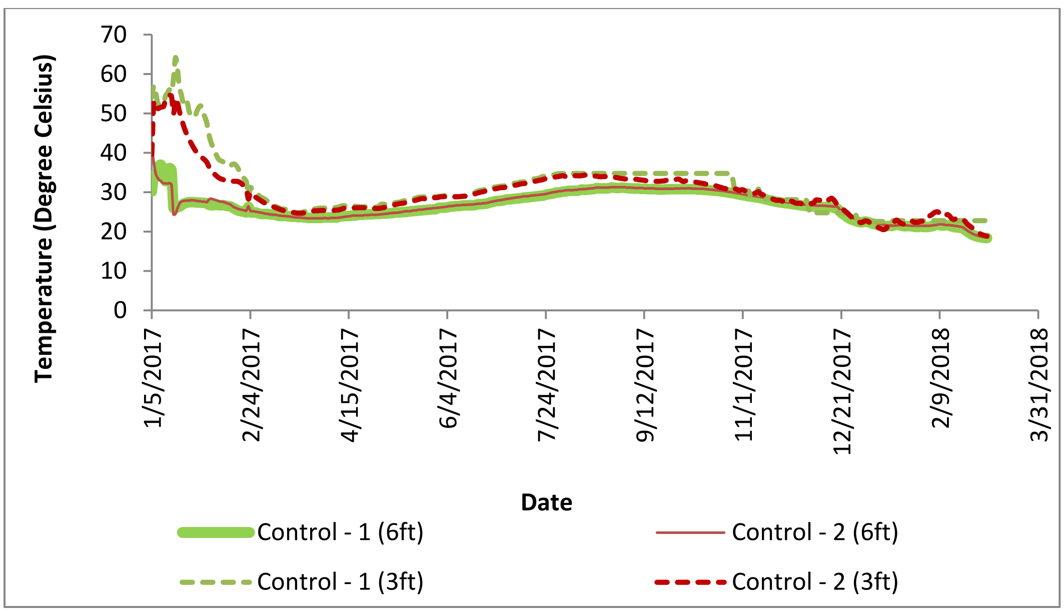


Figure 6-27 Temperature of waste in control cell

The average daily temperature in the control cell ranged from 24 °C – 64 °C in the first month of monitoring. It dropped to 26 °C in third month, and began increasing in May, 2017. The temperature at six feet was lower than that at three feet in both the biocell and control cells because the moisture content was higher at the six foot depth. Also, the three-foot depth was close to the lids of the cells, resulting in a higher temperature, as the metal lid absorbs the ambient temperature from outside the cells. To conclude, the temperatures of the two cells were indistinguishable after six months. The average daily temperature in the biocell in January, 2017, ranging from 15 °C – 36 °C, was significantly above the ambient air temperature during the winter months in Denton, which shows the initiation of biological activities inside the cell. In a study by Yazdani (2010), the temperature of waste in a digester cell was more than the ambient temperature due to the biological degradation of waste, as shown in Figure 6-29. In the filling phase, the temperature was very high (more than 70°C); in the anaerobic phase, it was reduced to almost 30 °C, which was close to the ambient temperature.

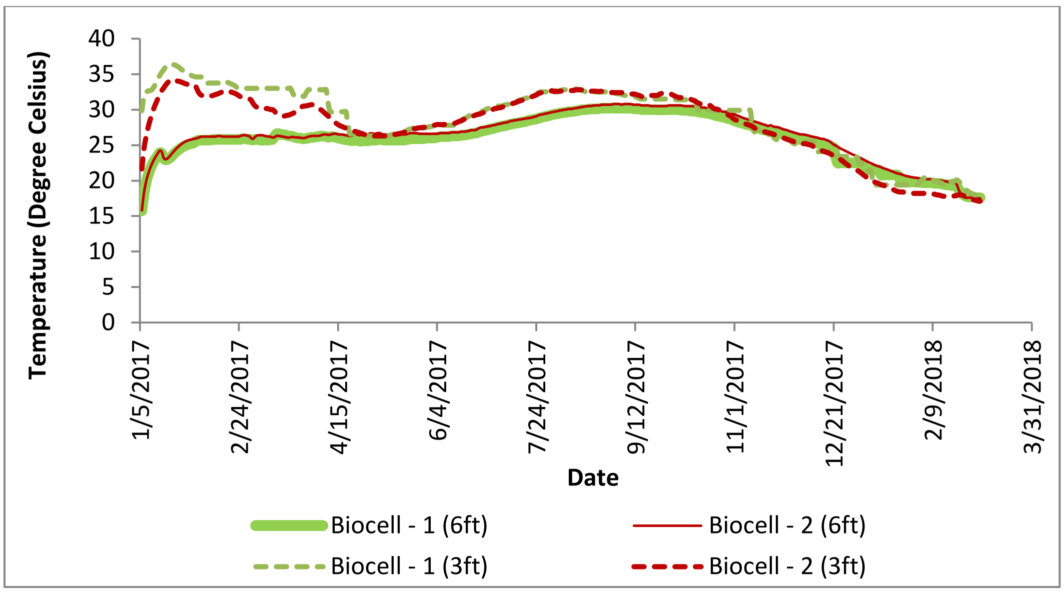


Figure 6-28 Temperature of waste in biocell

In the current study, the temperature inside the cells was higher than the ambient temperature, which indicated biological activities. After four months, the temperature was about 26 °C - 28 °C and continued increasing during the summer. After the middle of October, 2017, the temperature began to decrease. In January, 2018 it was 18 °C, which was higher than the ambient temperature during winter. According to several studies, the temperature in a landfill is expected to be higher than the atmospheric temperature because of the degradation of waste, which is an exothermic reaction (Christensen and Kjeldsen, 1989; Rees, 1980; Bingemer and Crutzen 1987).

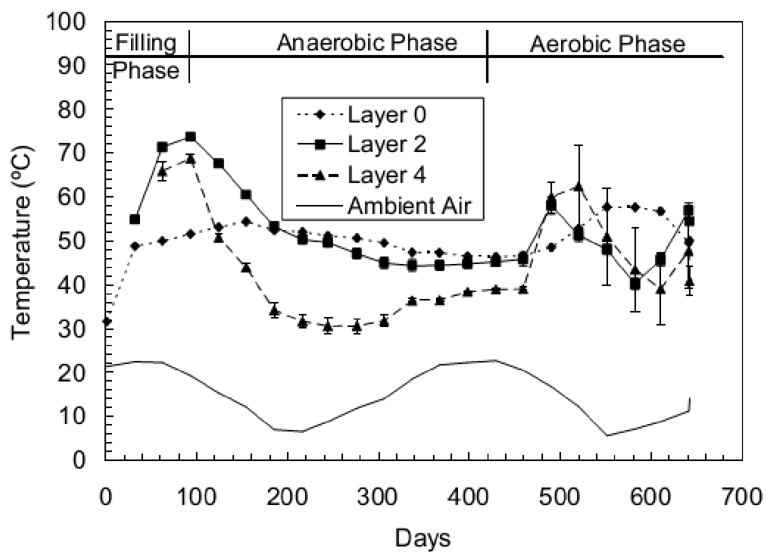


Figure 6-29 Digester cell average monthly waste temperature (Yazdani, 2010)

Temperature has a significant effect on methane production, as well as waste degradation, as has been observed in several studies (Christensen and Kjeldsen, 1989; Tchobanoglous et al., 1993; Buivid et al., 1981; and Khanal, 2008). According to Tchobanoglous et al. (1993), methane generation is significantly reduced at temperatures lower than 20 °C and higher than 70 °C. Buivid et al. (1981) found that the optimum temperature for methane production is 37 °C. A clear relationship between methane production and temperature was observed from the field scale data of this study. The amount of methane generated from each cell was plotted against the temperature of the respective cell, as shown in Figure 6-30. The highest methane production in both cells as observed during summer, 2017, when the ambient temperature was around 30 °C. This helped to increase the temperature in the control cell and biocell, which ultimately affected the methane production. From the results of both of the cells, it was clear that temperature exponentially affects the methane volume; however, this relationship cannot be verified, as the cells experienced only two winters and one summer.

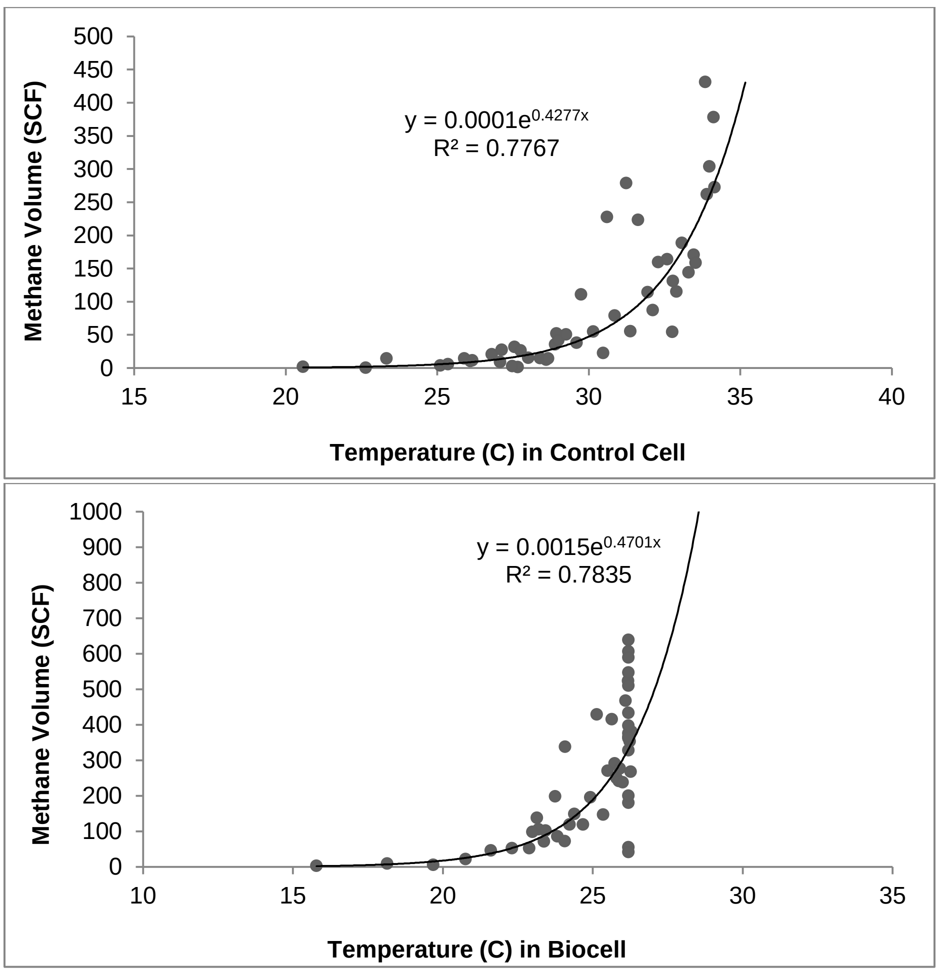


Figure 6-30 Effect of temperature on methane production

6.8 Decay Rate (k) of Waste in Field Study

Methane generation from a landfill depends on the methane generation potential (L_o) and methane generation rate factor (k) of the waste mass deposited in landfill. Most of the landfill gas generation models use these two parameters to estimate the theoretical methane production, which is typically modeled using the U.S. EPA's LandGem model (U.S. EPA 2005) in the following Eq. 6-1:

$$Q_M = k.L_o \sum_{i=0}^{n} \sum_{j=0}^{1} \frac{M_i}{10} e^{-k.t_{ij}} \quad \text{.......................................} \; .(6\text{-}1)$$

Where,

Q_M = Methane generation rate in the n^{th} year (m^3 yr^{-1}),

L_o = Methane generation potential (m^3 Mg^{-1}),

Mi = Mass of solid waste place in biocell in the i^{th} year (Mg),

k = Decay rate constant ($year^{-1}$),

j = Decimal year time increment,

t = Time (year)

The methane generation rate is also known as the decay rate, kinetic constant, or rate of degradation and is governed by several landfill-dependent variables, such as ambient temperature, precipitation, waste composition, moisture, landfill depth, availability of nutrients, and pH of leachate (Barlaz et al., 1990). Several studies have been conducted to estimate the decay rate, although it is difficult to measure since it depends on the conditions within the landfill cell such as moisture content, temperature and nutrient availability. An alternative term used to indicate the decay rate is "half-life" ($t_{1/2}$), which means the amount of time required for the conversion of the degradable organic waste to half of its initial mass. The half-life and rate constant of slowly and rapidly biodegradable wastes vary significantly. The relationship between k and $t_{1/2}$ is given in Eq. 6-2.

$$k = \frac{ln(2)}{t_{1/2}} \text{ ....(6-2)}$$

Once the field test cells were filled with the feedstock and inoculum, they were covered with the lids immediately so that most of the gas produced could be collected. No additional solid was added after the lid was placed on the box, which allowed for a straightforward analysis of decay rate. The cumulative methane generation from the field test cells was used to calculate a decay rate value, applying a standard first order methane gas production model. The methane volume was measured from the gas volume generated from the cells and methane content in the gas composition. To develop Eq. 6-3, which was used to estimate the decay rate, an integrated form of Eq. 6-1 was used (Barlaz, et al., 2010).

$$V = L_o M(1 - e^{-kt}) \text{ ....(6-3)}$$

Where V = Cumulative CH_4 collected once the lid was placed over the cells to time t (m^3),

M = Initial mass of solids placed in the cells (Mg),

L_o = Methane generation potential, m^3 Mg^{-1},

k = Decay rate constant ($year^{-1}$),

t = Time (year)

Using Eq. 6-3, the decay rate was calculated as presented in Table 6.3. The estimated decay rate of the biocell was considerably larger (1.32 $year^{-1}$) than the decay rate of the control cell (0.18 $year^{-1}$) and the values found in literature. Table 6.3 shows a comparison of the various decay rate constant values found in the literature with those from this study. The k values found in literature for bulk waste in US landfills varied from

0.003 to 0.21 $year^{-1}$ (USEPA, 2005). In a study by Hunte (2010), the kinetic rate constant for residential waste in Calgary was estimated to be 0.131 $year^{-1}$ and 0.146 $year^{-1}$ for the biocell he worked on. Yazdani (2010) conducted another study on green waste and horse manure in California, and found the decay rate to be 0.82 $year^{-1}$.

Table 6.3 Comparison of decay rate constants of current study with values reported in literature

Decay Rate Constant	Type of Landfill	Reference
k = 0.02 $year^{-1}$	Dry landfill receiving less than 20 inches of annual precipitation	EPA (2010)
k = 0.04 $year^{-1}$	Moderate landfill receiving between 20 and 40 inches of annual precipitation	EPA (2010)
k = 0.06 $year^{-1}$	Wet landfill receiving greater than 40 inches of annual precipitation	EPA (2010)
k = 0.12 $year^{-1}$	Bioreactor landfill operating as bioreactors where water is added until the moisture content reaches 40 percent moisture on a wet-weight basis	EPA (2010), Barlaz et al. (2010) and Tolaymat et al. (2010)
k = 0.052 $year^{-1}$	National Average, corresponding to a weighted average based on the share of waste received at each landfill type	EPA (2010)
k = 0.3 $year^{-1}$	Proposed for well-designed wet landfills in the USA	Faour et al. (2007)
k = 0.146 $year^{-1}$	Calgary Biocell	Hunte (2010)
k = 0.82 $year^{-1}$	Estimated for digester cell fed with green waste and horse manure	Yazdani (2010)
k = 1.32 $year^{-1}$	Biocell	This study
k = 0.18 $year^{-1}$	Control Cell	

Other researchers found decay rates of different waste components. Cruz and Barlaz (2010) conducted a study to find the decay rate for different waste components (i.e. paper, food waste, woods, textile, miscellaneous organics etc.), based on a study by Eleazar et al. (1997). Laboratory scale k values were measured for these waste components, based on the rates of degradation observed in laboratory-scale bioreactors and were converted to field scale. They found that the highest decay rate was for grass (0.6 $year^{-1}$,); food waste had the decay rate of 0.289 $year^{-1}$ (Eleazer et al. 1997; Cruz and Barlaz, 2010). The estimated decay rate, based on the methane production in this study, was higher than any of the decay rates found by previous researchers. A higher value of decay rate from the field biocell in this study indicated accelerated waste decomposition in the biocell. It was almost 11-fold and 33-fold the U.S. EPA bioreactor landfill default decay rate (k = 0.12 $year^{-1}$) and conventional landfill default decay rate (k = 0.04 $year^{-1}$), respectively.

6.9 Effect of Pig Manure on Methane Production from Field Scale Cells and Laboratory Simulated Reactors

Methane volumes in the field-scale biocell and control cell were compared with the laboratory scale study, and the results are shown in Figure 6-31 to Figure 6-34. Field data was converted to the same scale as the laboratory data for the comparison. The laboratory scale reactor with pig manure and sludge produced a considerable amount of methane after only 20 days. About 47 L/lb. of methane was generated by the reactor with pig manure in 233 days. In the field, suction was applied for the first time to extract the gas on day 43, when the methane content was reasonably high. The lag phase in the field scale biocell was almost two times that of the laboratory-scale reactor, but that is reasonable, considering the differences in the mass of feedstock. Only 2 lbs. of wet waste were used in the laboratory, whereas was almost 5 tons (10,000 lbs.) were used in the field. So even though the field test cells had almost 5,000 times more waste than the laboratory scale reactors, the lag phase was negligible. The difference in the duration of the lag phases in the field and laboratory might be due to the heterogeneity of the waste. Even though similar compositions of paper, food, yard, and textiles were used, as well as similar combinations of inoculum, the heterogeneity still exists among different types of paper, food, yard, and textile waste. In the field, about 53 L/lb. of methane was produced in 424 days, which was more than the cumulative methane volume produced from the biocell with pig manure in the laboratory. The volume of methane produced from the biocell in the field was similar to the volume of methane produced from the laboratory-scale MSW reactors with MnP (M9 and M10), as described in Chapter Four. Figure 6-31 shows a comparison of the cumulative methane volumes (L/lb.) of the biocell in the laboratory and field.

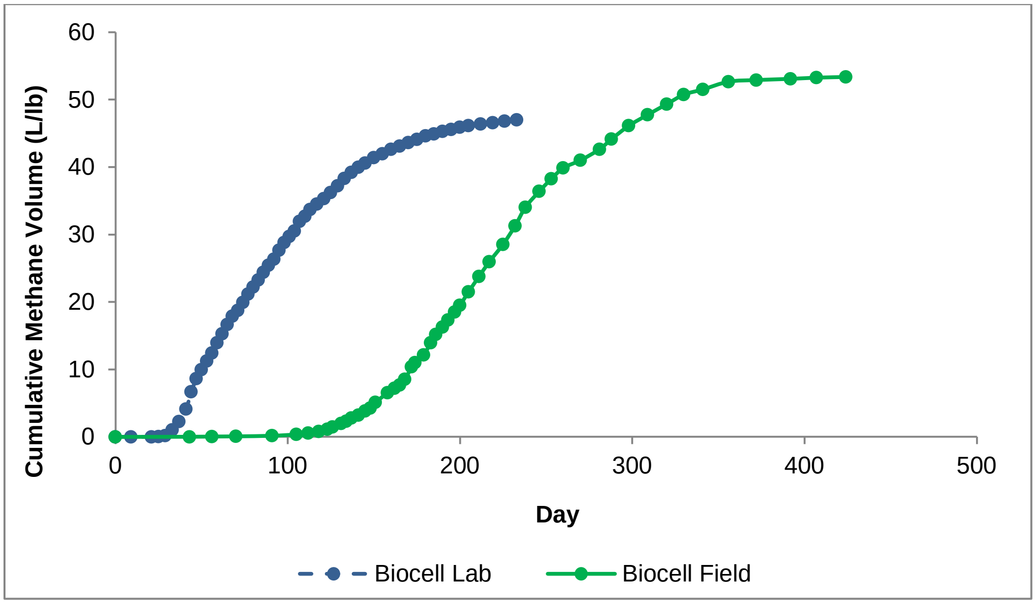


Figure 6-31 Comparison of cumulative methane volume (L/lb.) of biocell in lab and field

The rates of total methane generation or yield (mL/lb./day) in the laboratory-scale simulated MSW reactor with pig manure and the field biocell are shown in Figure 6-32. The peak value of the methane yield observed from the laboratory-scale biocell was 856 mL/lb./day on day 44, while the biocell in the field experienced two peaks of maximum methane generation rate: 459 mL/lb./day on day 172 and day 238. The reason behind the difference in the methane yield of field and laboratory biocells can be explained by the waste composition and mass. In the laboratory experiment, waste was separated and shredded, and only organic waste was used in reactors. But in the field, it was a challenge to sort about 5 tons of waste to take out all of the inorganics. Therefore, there was a significant difference in the heterogeneity of the waste in the field and in the laboratory.

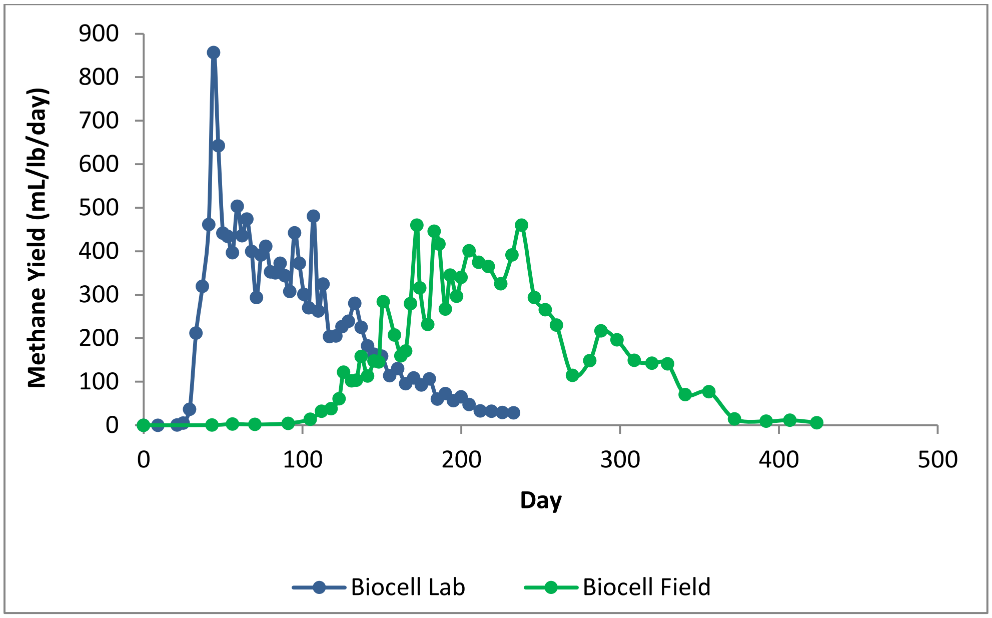


Figure 6-32 Comparison of methane yield (mL/lb./day) of biocell in lab and field

The field control cell also produced more methane than the laboratory-scale control reactor with MSW. The control cell in the laboratory produced only 2.45 L/lb. of methane in 113 days, then stopped production; the field-scale control cell produced 13.14 L/lb. methane in 425 days. The field- scale control cell produced almost six times more methane than the control reactor in the laboratory. The amount of feedstock plays an important role in methane production in the field. Figure 6-33 shows a comparison of the cumulative methane volume (L/lb.) in the control cells in the lab and field, where the methane yield in the field scale control cell was higher than that in the laboratory-scale MSW control reactor. In the laboratory, the highest methane yield was 70 mL/lb./day on day 110. The laboratory reactor experienced three peaks: on day 44, day 86, and day 110. The field control cell had one peak of 203 mL/lb./day on day 212. Figure 6-34 shows the comparison of methane yield (mL/lb./day) of the control cell in the laboratory and field.

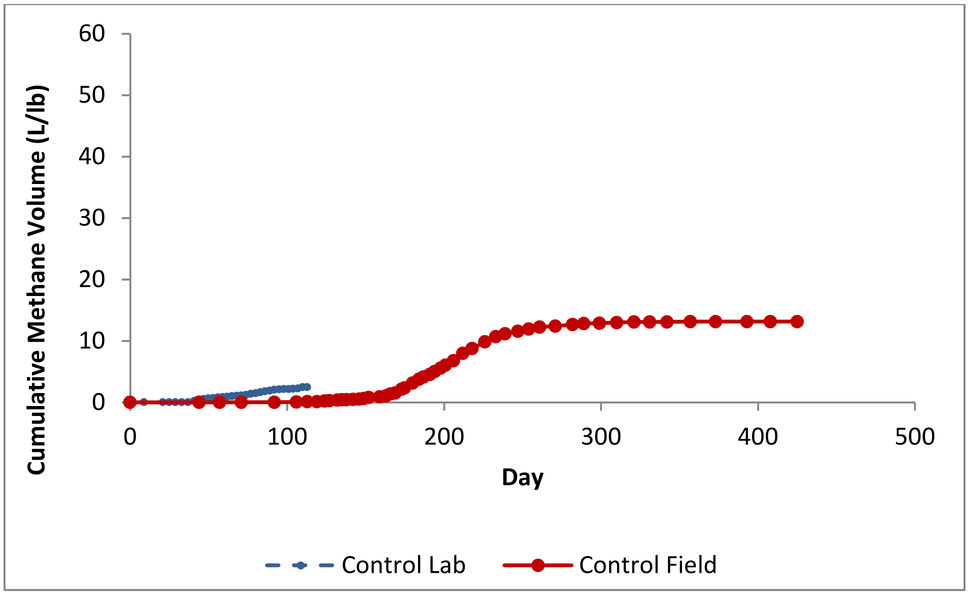


Figure 6-33 Comparison of cumulative methane volume (L/lb.) of control cell in lab and field

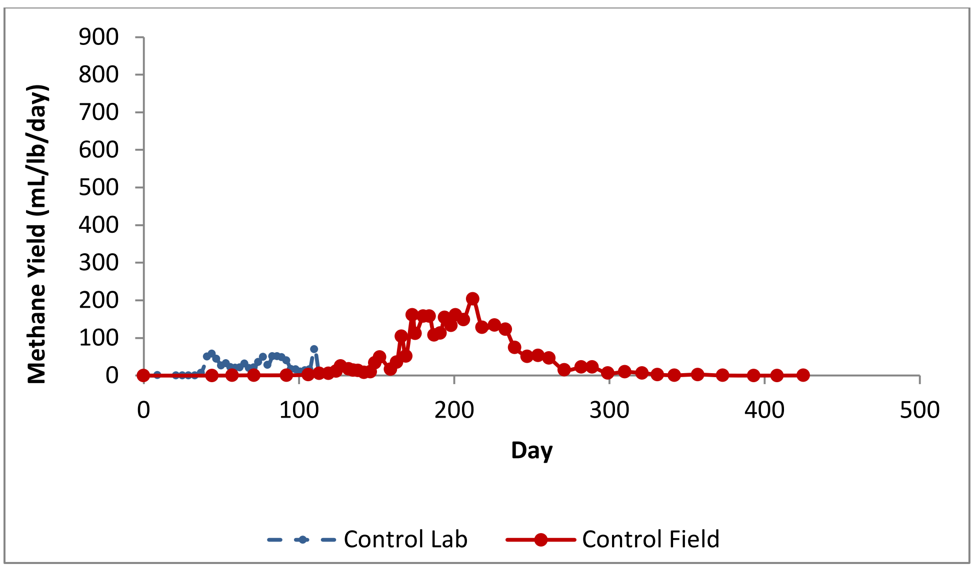


Figure 6-34 Comparison of methane yield (mL/lb./day) of control cell in lab and field

6.10 Summary

The process of enhancing methane production by the addition of pig manure in laboratory-scale experiment was converted to the field scale successfully in this study. The increased gas production from the manure-enhanced field-scale biocell confirmed the feasibility of adding pig manure to enhance methane production and waste decomposition. The biological enhancement of waste in the biocell, by adding pig manure, resulted in increasing the biodegradability of the waste. The biocell produced three times more methane than the control cell, similar to the observation in the laboratory-scale landfill simulation presented in Chapter Four. However, a lower amount of methane was observed in the field than in the laboratory experiments, due to the heterogeneity of the waste and lack of temperature control in the field. Based on the results and analyses of the field-scale study of the biocell, it can be concluded a sustainable waste management system can be implemented by operating a landfill as a biocell. It will increase the methane production and reduce the time required for the waste to decompose, hence reducing the need for new landfill space.

Chapter 7

Conclusions and Recommendations

7.1 Summary and Conclusion

This study focuses on the design and operation of a biocell for field application as part of a sustainable waste management system. Solid waste management is a challenge globally, and organic fractions of MSW are difficult to manage, as they are the main source of pollution and the largest constituents of the waste mass. Landfilling is the most common disposal technique used worldwide as it is the easiest and cheapest way to dispose of waste. The land required for landfilling, which has always been an issue, has led to the concept of a bioreactor o eliminate the problem of slow biodegradation of waste in a conventional landfill. A biocell is an advanced example of a bioreactor landfill, and is associated with increased waste degradation, faster waste stabilization, and a higher rate of gas generation. Enhanced waste degradation can be achieved by recirculating leachate in the waste mass, enhancing the leachate by adding nutrients and a buffer before recirculation, and adding inoculum such as manure, sludge, degraded waste, old leachate, etc. A biocell increases waste degradation, gas generation, and waste stabilization. The main features of a biocell operation are the addition of inoculum (sludge and manure) and recirculation of the collected leachate into the refuse mass to facilitate biodegradation. Due to the presence of microbes and moisture, the methane generation rate increases, with a higher gas generation yield over time. A number of studies have been conducted on laboratory-simulated landfill reactors to which sludge was added, but no study has been conducted to observe the effects of different types of manure on the degradation of MSW and food waste in a laboratory-simulated landfill. Moreover, the effects of adding manure to fresh MSW has not been studied in a field-scale landfill. This study focuses on a laboratory-scale study of the effects of manure and enzymes on the degradation of the organic fraction of MSW and food waste, then applies the best combination, in terms of methane production, to the field.

In the laboratory-scale study, fresh municipal solid wastes were collected from the working phase of the City of Denton Landfill. Food waste was collected from two sources: fruit and vegetable wastes were collected from Walmart, Denton, Texas, and food scrapings were collected from the University Center Cafeteria (Connection Café) at the University of Texas at Arlington. Five pairs of MSW reactors (M1 to M10) and four pairs of food waste reactors (F1 to F8) were built and monitored periodically to measure the gas volume and composition, as well as leachate quality and quantity. After 241 days of operation, the MSW reactors were dismantled; after 579 days of operation, the food waste reactors were dismantled. Once the laboratory tests on the MSW reactors were complete, the field-scale biocell and control cell were installed in in the City of Denton Landfill and monitored for more than one year, until the field test cells stopped producing methane. The results from the laboratory-scale study and field tests are summarized as follows:

- MSW samples and food waste samples, collected from the working face of the City of Denton Landfill, Walmart, Denton, Texas and University Center Cafeteria (Connection Café) at the University of Texas at Arlington were sorted to determine their physical composition. On average, the MSW sample was composed of 34% paper, 19% plastic, 13% food waste, 8% textile, 2% styrofoam and sponge, 9% yard and wood waste, 3% metals, 2% glass, 4% construction debris and 6% others (soils and fines). The food waste consisted of 49.85% fruits and vegetables, 30.15% grain products, 9.98% meat and seafood, and 2.02% dairy products.

- The MSW was sorted manually to separate the degradable products (paper, food waste, yard waste and textile) to be fed into the MSW reactors. Based on the degradable waste composition of the collected MSW from the City of Denton Landfill, the composition of the reactors'' feedstock was determined as 50% paper, 20% food waste, 15% textiles, and 15% yard and wood waste. The national food waste composition was used for the food waste reactors, which included fruits and vegetables (50%), meat and seafood (20%), grain products (20%), and dairy products (10%).
- The average moisture content of the fresh MSW was 26.7% on wet weight basis; the average moisture content of fresh food waste was about 76.55% on wet weight basis. The moisture content of sludge, cow manure, pig manure and horse manure was found to be 75.67%, 2%, 5%, and 36%, respectively.
- The average amount of volatile solids in the MSW feedstock in the reactors was about 85.9%, and in food waste feedstock was about 92.16%. The moisture content and volatile content for the duplicate reactors were found to be similar because of the similar composition of waste in the reactors.
- Cow manure, pig manure, horse manure, and sludge were used as inoculum in this study. The pH of cow manure, pig manure, horse manure, and sludge was found to be 8.95, 7.81, 7.69, and 8.37, respectively.
- Five pairs of MSW reactors (M1 to M10) were seeded with an organic fraction of MSW, and four pairs of food waste reactors (F1 to F8) were seeded with food waste. In the control reactors (M1, M2, F1 and F2), only organic fractions of MSW and food waste were used, respectively. Six percent of cow manure, pig manure, and horse manure was used in three pairs of MSW reactors (M3 to M8) and three pairs of food waste reactors (F3 to F8). Manganese peroxide (MnP)

was used in one pair of MSW reactors (M9 and M10). Each MSW reactor was seeded with 2 lbs. of municipal solid waste, and each food waste reactor was seeded with 4 lbs. of food waste, then kept in an environmental growth chamber at 37 ^{o}C.

- Gas composition and flow from the reactors were observed periodically. The highest methane contents seen were 66.20% and 66.10% in reactors M5 and M6, respectively, with pig manure, followed by reactors M3 and M4 with cow manure. Reactors with MnP had a methane content of 64.1% in M9 and 62.6% in M10. The lowest methane content was seen in the control reactors (M1 and M2), which was 52.3 and 60.1%, respectively.
- The food waste reactors with cow manure (F3 and F4) produced more than 70% methane almost continuously for 150 days. The highest methane content was about 76.10% in reactor F3 on day 198. All of the food waste reactors except F8 achieved more than 70% methane during operation, which is significantly higher than landfill methane content and similar to the methane content in anaerobic digester.
- Reactors with MnP (M9 and M10) produced the highest amount of gas, which was about 100.6 L/lb. and 105.1 L/lb., respectively, in 233 days. The control reactors (M1 and M2) produced only 10.3 L/lb. and 24.1 L/lb. of gas, respectively. Reactors with pig manure (M5 and M6) produced the second highest amount of gas, which amounted to 85.4 L/lb. and 83.3 L/lb., respectively, in 233 days.
- Reactors M9 and M10 generated about 54.5 L/lb. and 52.03 L/lb. of methane, respectively, in 233 days, which was the highest among all of the MSW reactors **and** almost 22 times more than control reactor, M1. Reactors with pig manure (M5 and M6) also performed well in terms of methane production from the

organic fraction of MSW, amounting to about 47 L/lb. and 46 L/lb., respectively in 233 days. Although the reactors with pig manure generated a slightly lower amount of methane than the reactors with MnP for the organic fraction of MSW, pig manure is much more cost effective than MnP for application to a large field scale.

- Food waste reactors with cow manure (F3 and F4) produced significant amounts of gas (68.5 L/lb. and 72 L/lb. respectively) in their lifetime of about 364 days, while MSW waste reactors with cow manure (M3 and M4) produced only 53.6 L/lb. and 13.2 L/lb. gas, respectively, in 233 days.
- Among the food waste reactors, the highest methane volume was generated from reactor F4 (43.3 L/lb.) which was close to the volume of methane produced in reactor M6, with pig manure. The lowest amount of methane was produced from food waste reactor F5 (only 12.7 L/lb.). Control food waste reactors (F1 and F2) also produced significant amounts of methane (about 15.9 L/lb. and 20 L/lb., respectively), while the control MSW reactors produced much less (2.5 L/lb. and 8.2 L/lb.). The food waste reactors generated a significant amount of methane, but it took more than twice the amount of time of the MSW reactors.
- The quality and quantity of leachate produced from the MSW and food waste reactors were also measured. Initially, the pH of leachate of all the MSW reactors was less than 7 due to the accumulation of acid during the acidogenic phase. The pH of all of the MSW reactors exceeded 6 within 50 days of operation and 7 within 110 days of operation. After that, all of the MSW reactors were stabilized with pH values between 7 and 8.
- A significant drop of pH was seen in the food waste reactors throughout the initial monitoring period, due to the accumulation of excessive volatile fatty acids

(VFA). For all the MSW reactors, it took about 110 days to reach the methanogenic phase; in the food waste reactors, only reactor B3 was able to reach more than 7 after 110 days. It took almost 370 days for all the food waste reactors to reach the methanogenic phase.

- The COD and BOD of the leachate in the MSW reactors and food waste reactors were measured on a monthly basis to determine the level of degradation of waste inside the reactors. The COD and BOD curves of leachate showed similar trends for both the MSW and food waste reactors.
- The ratio of BOD to COD in leachate is an indicator of waste degradation, and it decreases with time, along with waste degradation. Initially, the BOD/COD ratio of all of the MSW reactors ranged from 0.7 to 0.6, indicating the presence of biodegradable materials in the MSW reactors. The BOD/COD for all of the MSW reactors, except M2, at the end of the monitoring period (about seven months) was around 0.06.
- At the beginning, the BOD/COD ratio for all the food reactors was about 0.75 to 0.64, similar to that of the MSW reactors. BOD/COD of all of the food waste reactors fluctuated rapidly until the 5^{th} month, except the reactors with cow manure (F3 and F4), and the value was always more than 0.6. Even after 19 months of the study, none of the food waste reactors were able to achieve a BOD/COD of 0.06, as the MSW reactors had.
- At the end of monitoring, the laboratory scale reactors were dismantled to measure the waste characteristics such as weight loss, settlement, moisture content, volatile solids, etc. after the biodegradation. The highest percentage of weight loss and settlement were observed in MSW reactors with MnP (M9 and M10), followed by MSW reactors with pig manure (M5 and M6). Among the food

waste reactors, the highest amount of waste loss was seen in reactor F2, followed by reactor F7. The highest settlement was observed in reactor F3, followed by F4, where gas production was the highest.

- The initial moisture content was similar in all reactors, but the final content varied because of different levels of degradation. The moisture content of most of the degraded food waste samples was higher than the initial moisture content, except for reactors F3, F4, and F5. Lowest moisture content was observed in reactor F4, followed by F3.
- The percent reduction of volatile solids in the waste is positively correlated with the total methane production from the waste. In other words, the more the volatile solids are reduced, the more methane will be generated. Among the MSW reactors, volatile solid reduction after the degradation of waste was observed to be highest for the MSW reactors with pig manure and MnP, and they generated the highest amount of methane. Food waste reactors showed similar results.
- Based on the laboratory scale landfill simulation, it can be concluded that pig manure and cow manure can be used as effective inoculum for MSW and food waste, respectively. Although the reactors with MnP (M9 and M10) produced the most methane, it may not be applicable to field use because of its very high price.
- After the laboratory scale study of MSW reactors was complete, two field-scale test cells (biocell and control cell) were installed the in City of Denton Landfill and were monitored for 14 months. Both cells were well equipped with leachate and gas management systems, as well as an automated system for monitoring the temperature and moisture within the cells. Pig manure was used in the biocell, where the control was simulated as a bioreactor landfill.

- After 145 days of operation, the methane content of the gas in the biocell reached 45.8%; in the control cell, it reached 45.4% on day 212.
- The cumulative gas volume in the biocell during 14 months of operation was about 30,336 standard cubic feet; in control cell, it was 15,553 standard cubic feet. Methane production was also higher in the biocell (12,437 SCF) than in the control cell (4,644 SCF). The average gas flow rate per minute was about 16.7 standard cubic feet in the biocell and 8.9 in the control cell.
- The peak value of methane yield observed from the lab-scale biocell was 856 mL/lb./day on day 44, while in the field, it was the highest (459 mL/lb./day) on day 172 and day 238. The field biocell produced 53.4 L/lb. of methane in 424 days of operation, while the lab biocell produced almost 47 L/lb. of methane in 233 days.
- The field control cell produced almost six times more methane than the lab. The control cell in the lab produced only 2.45 L/lb. methane in 113 days, then stopped producing. The field-scale control cell produced 13.14 L/lb. of methane in 425 days.
- Leachate production in the control cell was higher than in the biocell, even though the moisture content was higher in biocell due to the addition of sludge, which has a moisture content of about 72%.
- Throughout the monitoring period, the pH of the leachate in the biocell remained above 7 due to the addition of sludge, which has an alkaline pH. In the control cell, the pH dropped below 7 in the 2nd month, but began increasing when the methane content went up. The amount of leachate generated was higher in the control cell than in the biocell, which supports the volume of methane generated.

- Based on sensor data, the daily average gravimetric moisture content in the biocell ranged from 34% to 64% in wet weight basis; in the control cell, it ranged from 32% to 60% in wet weight basis.
- The average daily temperature in the biocell and control cell ranged from 15 °C – 36 °C and 24 °C - 64 °C, respectively, in January 2017, during the filling phase. After January, the average daily temperatures in the biocell and control cell were about 26 °C - 29 °C.
- The estimated decay rate of the biocell was considerably larger (1.32 $year^{-1}$) than the decay rate of the control cell (0.18 $year^{-1}$) and other values found in literature (0.003 to 0.21 $year^{-1}$).
- From the results of landfill biogas quality and quantity, and leachate quality, it can be concluded that pig manure enhanced the MSW biodegradation in both the laboratory-scale landfill simulation and field application done in this study. By operating landfills as biocells, sustainable waste management systems can be achieved with enhanced methane production and waste decomposition.

7.2 Recommendation for Future Studies

Based on the observation and experience gained from experiments of the current study, several recommendations are proposed for future studies.

- The study presented both laboratory-scale and field-scale biocell scenarios; however, it is recommended that actual landfill conditions be applied in future studies.
- As pig manure and cow manure performed as the best inoculum for fresh organic MSW and food waste, respectively, it would be helpful to vary the combinations of them in laboratory simulation to find the optimum blend.

- Food waste’s potential for producing methane is excellent if the lag phase can be shortened. A new study could reduce the lag phase, which would eventually enhance the methane production.
- The effect of cellulose and hemicellulose on degrading enzymes in fresh waste, could be studied, although fresh waste is easy to degrade naturally.
- Further research is recommended to study the effects of temperature and moisture on methane production by extending the monitoring period.
- The effect of the manure’s age on methane production was not addressed in this study because of time limitations. Future research would be helpful in determining the optimum age of manure.
- The use of an optimum combination of anaerobically digested sludge from waste water treatment plants could be demonstrated in future studies.
- Future study is recommended on the effect of the waste composition on methane production in developing countries, where the organic waste content is significant for successful biocell implementation.
- A life cycle analysis (LCA) would be helpful on operating a landfill as a biocell to determine the environmental impacts of biocell operation compared to other waste disposal techniques.

Appendix

Type of Identified Bacteria

B1 (Control Cell)

Month	Type of identified bacteria		Bacterial Species
	Methanogenic	Other	
January	1	4	Polynucleobacter necessarius subsp. necessarius (strain STIR1), Escherichia coli O157:H7, Shewanella frigidimarina (strain NCIMB 400), Brucella abortus biovar 1 (strain 9-941), **Methanococcus voltae**
March	0	9	Bacillus subtilis (strain 168), Treponema pallidum (strain Nichols), Klebsiella pneumoniae, Enterobacter cloacae, Haemophilus influenzae (strain ATCC 51907 / DSM 11121 / KW20 / Rd), Bacillus halodurans (strain ATCC BAA-125 / DSM 18197 / FERM 7344 / JCM 9153 / C-125), Shigella flexneri, Rhizobium leguminosarum bv. viciae (strain 3841), Klebsiella pneumoniae
April	1	2	Hamiltonella defensa subsp. Acyrthosiphon pisum (strain 5AT), Koribacter versatilis (strain Ellin345), **Methanocaldococcus jannaschii (strain ATCC 43067 / DSM 2661 / JAL-1 / JCM 10045 / NBRC 100440)**
June	1	3	Paracoccus denitrificans (strain Pd 1222), Escherichia coli, Caulobacter crescentus (strain ATCC 19089 / CB15), **Methanosarcina mazei (strain ATCC BAA-159 / DSM 3647 / Goe1 / Go1 / JCM 11833 / OCM 88)**
October	3	3	Paraburkholderia xenovorans (strain LB400), Geobacter metallireducens (strain GS-15 / ATCC 53774 / DSM 7210), Thermobifida fusca (strain YX), **Bacillus subtilis (strain 168), Methanococcus voltae, Methanosarcina mazei (strain ATCC BAA-159 / DSM 3647 / Goe1 / Go1 / JCM 11833 / OCM 88)**

B2 (Biocell)

Month	Type of identified bacteria		Bacterial Species
	Methanogenic	Other	
January	3	3	Thermotoga neapolitana (strain ATCC 49049 / DSM 4359 / NS-E), Helicobacter pylori (strain HPAG1), Vibrio parahaemolyticus serotype O3:K6 (strain RIMD 2210633), **Escherichia coli (strain K12), Methanocaldococcus jannaschii (strain ATCC 43067 / DSM 2661 / JAL-1 / JCM 10045 / NBRC 100440), Methanococcus voltae**
February	1	1	Lysinibacillus sphaericus (strain C3-41), **Methanocaldococcus jannaschii (strain ATCC 43067 / DSM 2661 / JAL-1 / JCM 10045 / NBRC 100440)**
April	2	3	Agrobacterium vitis (strain S4 / ATCC BAA-846), Bacillus subtilis (strain 168), Bacillus halodurans (strain ATCC BAA-125 / DSM 18197 / FERM 7344 / JCM 9153 / C-125), **Methanocaldococcus jannaschii (strain ATCC 43067 / DSM 2661 / JAL-1 / JCM 10045 / NBRC 100440), Methanosarcina mazei (strain ATCC BAA-159 / DSM 3647 / Goe1 / Go1 / JCM 11833 / OCM 88)**
July	3	13	Mycobacterium avium (strain 104), Cloacimonas acidaminovorans (strain Evry), Borrelia hermsii (strain HS1 / DAH), Clostridium kluyveri (strain NBRC 12016), Shigella dysenteriae, Bacillus subtilis (strain 168), Escherichia coli O157:H7, Treponema pallidum (strain Nichols), Clostridium symbiosum, Enterobacter cloacae, Bacillus halodurans (strain ATCC BAA-125 / DSM 18197 / FERM 7344 / JCM 9153 / C-125), Magnetospirillum magneticum (strain AMB-1 / ATCC 700264), Klebsiella pneumoniae, **Escherichia coli (strain K12), Methanocaldococcus jannaschii (strain ATCC 43067 / DSM 2661 / JAL-1 / JCM 10045 / NBRC 100440), Methylobacterium extorquens (strain ATCC 14718 / DSM 1338 / JCM 2805 / NCIMB 9133 / AM1)**
October	3	7	Burkholderia mallei (strain NCTC 10229), Shewanella loihica (strain ATCC BAA-1088 / PV-4), Sulfurovum sp. (strain NBC37-1), Sinorhizobium medicae (strain WSM419), Bacillus subtilis (strain 168), Escherichia coli O157:H7, Bacillus halodurans (strain ATCC BAA-125 / DSM 18197 / FERM 7344 / JCM 9153 / C-125), **Methanocaldococcus jannaschii (strain ATCC 43067 / DSM 2661 / JAL-1 / JCM 10045 / NBRC 100440), Methanococcus voltae, Methylobacterium extorquens (strain ATCC 14718 / DSM 1338 / JCM 2805 / NCIMB 9133 / AM1)**

References

Alam, M. J. B., 2017. "Evaluation of Plant Root on the Performance of Evapotranspiration (ET) Cover System". Ph.D. Dissertation, University of Texas at Arlington, October 2017.

Alam, M. Z., 2016. “Moisture Distribution Efficiency and Performance Evaluations of Bioreactor Landfill Operations.” Ph.D. Dissertation, University of Texas at Arlington, November 2016.

Alkaabi, S.; Van Geel, P. J.; and Warith, M. A., 2009. “Effect of Saline Water and Sludge Addition on Biodegradation of Municipal Solid Waste in Bioreactor Landfills.” Waste Management and Research 27 (1), 59-69.

APHA., 2005. “Standard methods: For the examination of water and wastewater”. Edited by A. D. Eaton, L. S. Clesceri, E. W. Rice and A. E. Greenberg. 21st ed. Washington D.C.: American Public Health Association, American Water Works Association, Water Environment Federation.

Bakas, I.; Andersen, F.M.; and Larsen, H. V., 2011. “Projections of Municipal Waste Management and Greenhouse Gases”, ETC/SCP working paper, European Topic Centre on Resource and Waste Management.

Barlaz, M. A.; Schaefer, D. M.; and Ham, R. K., 1989. “Bacterial Population Development and Chemical Characteristics of Refuse Decomposition in a Simulated Sanitary Landfill.” Appl. Env. Microbiol, 55, 1, p. 55 - 65.

Barlaz, M.A.; Ham, R.K.; and Schaefer, D.M., 1990. “Methane Production from Municipal Refuse: A Review of Enhancement Techniques and Microbial Dynamics.” Crit. Rev. Environ. Contr. 19 (6), 557–584.

Barrett, A., and Lawlor, J., 1995. “The Economics of Waste Management in Ireland.” Economic and Social Research Institute, Dublin. 129 pp.

Bartholameuz, E. M, 2015. “Performance Enhancement of Aerobic Landfill Bioreactors.” PhD Dissertation, University of Calgary.

Białowiec, A., 2011. “Chapter 1: Hazardous Emissions from Municipal Solid Waste Landfills.” Contemporary Problems of Management and Environmental Protection, No. 9, 2011. “Some Aspects of Environmental Impact of Waste Dumps.“

Bisaria V.S. *Bioprocessing of agro-residues to glucose and chemicals. In Bioconversion of Waste Materials to Industrial Products*. Edited by: Martin AM. London: Elsevier; 1991:210–213.

Blakey, N. C.; Bradshaw, K.; Reynolds, P.; and Knox, K., 1997. “Bioreactor Landfill - A Field Trial of Accelerated Waste Stabilization.” Proceedings from Sardinia 97, Sixth International Landfill Symposium, Vol. I, S. Margherita di Pula, Cagliari, Italy, October 13-17, 375-386.

Breitmeyer, R.J. and Benson, C.H., “Measurement of Unsaturated Hydraulic Properties of Municipal Solid Waste.” Geo-frontiers 2011: Advances in Geotechnical Engineering, ASCE (2011), pp. 1433-1442.

Cedergren, H.R. (1989), “Seepage, Drainage, and Flow Nets”, Third Edition, John Wiley & Sons, New York, p. 465.

Christensen, T. H. and Kjeldsen, P., 1989. “Basic Biochemical Processes in Landfills.” In: Sanitary Landfilling: Process, Technology, and Environmental Impact. Academic Press, New, pp. 29-49.

Cirne, D.G.; Agbor, V.B.; and Bjornsson, L., 2008. “Enhanced Solubilization of the Residual Fraction of Municipal Solid Waste.” Water Sci. Technol. 57 (7), pp 995–1000.

Crawford, J.F. and Smith, P.G., 1985. Landfill Technology, Butterworth, London.

DeVries, B., 2016. "Hydraulic Performance Evaluation of Evapotranspiration Cover Systems". Ph.D. Dissertation, University of Texas at Arlington, November 2016.

Duan, N., Dong, B., Wu, B., Dai, X., 2012. High-Solid Anaerobic Digestion of Sewage Sludge under Mesophilic Conditions: Feasibility Study. Bioresour. Technol. 104,150 - 156.

EEA (European Environment Agency) 2003. “Europe’s Environment: The Third Assessment. Environmental Assessment Rept. No. 10 EEA, Copenhagen, Ch 7, 151-164.

Faour, A.; Reinhart, D. R.; and You, H., 2007. “First-Order Kinetic Gas Generation Model Parameters for Wet Landfills.” Waste Manage., vol. 277, pp 946–953.

Gallert, C.; Bauer, S.; and Winter, J., 1998. “Effect of Ammonia on the Anaerobic Degradation of Protein by a Mesophilic and Thermophilic Biowaste Population.” Applied Microbiology and Biotechnology, vol. 50, pp 495-501.

Guendouz, J., Buffiere, P., Cacho, J., Carrere, M., Delgenes, J.P., 20 08. High-solids Anaerobic Digestion: Comparison of Three Pilot Scales. Water Sci. Technol. 58, 1757-1763.

Hettiaratchi, J.P.A., 2006. Bio-Cell Project, *http://www.ucalgary.ca/EN/resrch_civil/bio-cell-project/Hettiaratchi-bio-cell-project.htm*.

Hettiaratchi, J.P.A.; Jayasinghea, P.A.; Bartholameuza, E.M.; and Kumarb, S., 2014. “Waste Degradation and Gas Production with Enzymatic Enhancement in Anaerobic and Aerobic Landfill Bioreactors.” Bioresource Technology, Vol. 159, May 2014, pp 433–436.

Higuchi, T., 2004. “Microbial Degradation of Lignin: Role of Lignin Peroxidase, Manganese Peroxidase and Laccase.” Proceedings of the Japan Academy (Series B80), pp 204-214.

Hoornweg, Daniel; Perinaz Bhada-Tata; and Chris Kennedy, 2013. "Waste Production must Peak this Century." Macmillan Publishers Limited.

http://www.unep.or.jp/ietc/publications/spc/State_of_waste.

https://www.usclimatedata.com/climate/denton/texas/united-states/.

Hunte, A. C., 2012. "Performance of a Full Scale Bioreactor Landfill." PhD Thesis. Department of Civil Engineering, University of Calgary.

Hunte, C.; Hettiaratchi, P.; Meegoda, N.J.; and Hettiarachchi, C.H., 2007. "Settlement of Bioreactor Landfills During Filling Operation." ASCE Geotechnical Special Publication #152, GeoDenver2007, ISBN # 0784408971.

Intergovernmental Panel on Climate Change (IPCC), 1996. "Good Practice Guidance and Uncertainty Management in National Greenhouse Gas Inventories." *http://www.ipccnggip.iges.or.jp/public/gp/english/5_waste.pdf.*

Jayasinghe, P.A.; Hettiaratchi, J.P.A.; Mehrotra, A.K; and Kumar, S., 2011. "Effect of Enzyme Additions on Methane Production and Lignin Degradation of Landfilled Sample of Municipal Solid Waste." Bioresource Technology, vol. 102 (2011), pp 4633–4637.

Jiang Y.; Heaven S.; and Banks C. J., 2012. "Strategies for Stable Anaerobic Digestion of Vegetable Waste." Renewable Energy, vol. 44(2012), pp 206-214.

Johannessen, L.M. 1999. "Guidance Note on Recuperation of Landfill Gas from Municipal Solid Waste Landfills." Working Paper Series, Urban Development, Division Urban Waste Management Thematic Group, World Bank.

Karanjekar, R. V., 2012. "An Improved Model for Predicting Methane Emissions from Landfills Based on Rainfall, Ambient Temperature and Waste Composition." PhD Dissertation, the University of Texas at Arlington.

Karanjekar, R., 2012. “An Improved Model For Predicting Methane Emissions From Landfills Based On Rainfall, Ambient Temperature And Waste Composition.” Ph.D. Dissertation, University of Texas at Arlington, June 2012.

Kayhanian, M., 1999. “Ammonia Inhibition in High-Solids Biogasification: An Overview and Practical Solutions.” Environmental Technology, vol. 20, pp 355 - 365.

Khanal, Samir Kumar. *Anaerobic Biotechnology for Bioenergy Production.* Wiley-Blackwell, 2008.

Knox, K. and Gronow, J.R., 1995. “Pilot Scale Study of Denitrification and Contaminant Flushing during Prolonged Leachate Recirculation.” Proceedings Sardinia '95, Fifth International Landfill Symposium, S. Margherita di Pula, Cagliari, Italy.

Kotsyurbenko, O. R., Chin, K.-J., Glagolev, M. V., Stubner, S., Simankova, M. V., Nozhevnikova, A. N. and Conrad, R., 2004. “Acetoclastic and Hydrogenotrophic Methane Production and Methanogenic Populations in an Acidic West-Siberian Peat Bog.” Environmental Microbiology, vol. 6, pp 1159-1173.

Lagerkvist, A. and Chen, H., 1993. “Control of Two Step Anaerobic Degradation of Municipal Solid Waste (MSW) by Enzyme Addition.” Water Science and Technology, vol. 27 (2), pp 47-56.

Liu, G.; Zhang, R.; El-Mashad, H. M.; and Dong, R., 2009. “Effect of Feed to Inoculum Ratios on Biogas Yields of Food and Green Wastes.” Bioresource Technology, vol. 100, no. 21, pp. 5103–5108.

Macias-Corral, M.; Samani, Z.; Hanson, A.; Smith, G.; Funk, P.; Yu, H. and Longworth, J., 2008. “Anaerobic Digestion of Municipal Solid Waste and Agricultural Waste and the Effect of Co-Digestion with Dairy Cow Manure”. Bioresource Technology 99: 8288–8293.

Martinez, A.T.; Speranza, M.; Ruiz-Duenas, F. J.; Ferreira, P.; Camarero, S.; Guillen, F.; Martinez, M. J.; Gutierrez, A.; and Rio, J. C., 2005. "Biodegradation of Lignocellulosics: Microbial, Chemical, and Enzymatic Aspects of the Fungal Attach of Lignin". International Microbiology 8 (3), 195-204.

McCarty, P. L., 1964. "Anaerobic Waste Treatment Fundamentals." III. Publ. Wks, vol. 95, pp 91-94.

Meegoda, J. K.; Bhuvaneshwari, S.; Hettiaratchi, P A.; and Hettiarachchi, H., 2013. "A Comprehensive Model for Anaerobic Degradation in Bio-Reactor Landfills." Seventh International Conference on Case Histories in Geotechnical Engineering.

Merz, R.C. and Stone, R., 1962. "Landfill Settlement Rates." Public Works, Vol. 93, No. 9, pp.103–106, 210–212.

Murphy, R.J., 1993. "Optimization of Landfill Mining." Technical Report, Center for Solid and Hazardous Waste Management and the Collier County Government of Naples, Florida.

Nikiema, J.; Brzenzinski, R.; and Heitz, M., 2007. "Elimination of Methane Generated from Landfills by Biofiltration: A review." Review of Environmental Science and Biotechnology, vol. 6, pp 61-284.

Pacey, J.; Yazdani, R.; Reinhart, D.; Morck, R.; and Augenstein, D., 1999. "The Bioreactor Landfill: An Innovation in Solid Waste Management." Solid Waste Association of North America, Silver Springs, Maryland.

Read, A.D.; Hudgins, M.; Harperc S.; Phillips P.; and Morrise, J., 2001. "The Successful Demonstration of Aerobic Landfilling: The Potential for a More Sustainable Solid Waste Management Approach." Resources, Conservation and Recycling, vol. 32, pp 115-146.

ReFED, 2016. “A Roadmap to Reduce U.S. Food Waste by 20 percent, Retrieved March 31, 2016, from *http://www.refed.com/downloads/ReFED_Report_2016.pdf.*

Reinhart, D. R.; McCreanor, P. T.; and Townsend, T., 2002. “The Bioreactor Landfill: Its Status and Future.” Waste Management and Research, vol. 20 (2), pp 172-186.

Reinhart, D.R. and Townsend, T.G., 1998. *Landfill Bioreactor Design and Operation*. Lewis Publisher, New York.

Sapkota, A., 2017. “Effect of Manure and Enzyme on the Degradation of Organic Fraction of Municipal Solid Waste in Biocell.” MS Thesis, University of Texas at Arlington, Spring, 2017.

Shao, L.; He, P. J.; Hua, Z.; Yu, X.; Li-Guo, J., 2005. “Methanogenesis Acceleration of Fresh Landfilled Waste by Micraeration." Journal of Environmental Sciences, vol. 17(3), pp 371-374.

Shihada, H.; Hossain, M.; Kemler, V.; and Dugger, D., 2013. ”Estimating Moisture Content of Landfilled Municipal Solid Waste without Drilling: Innovative Approach.” J. Hazard. Toxic Radioact. Waste, vol. 17(4), pp 317–330.

Solid Waste Association of North America, 2000. “Bioreactor Landfills: SWANA’s Comment.”

Sperling, T., 2009. “Personal Communication, Landfill Fire at Municipal Landfills”. Sperling Hansen Associates Inc.

Stegmann, R. and Spendlin, H. H., 1989. *Enhancement of Degradation: German Experiences. In Sanitary Landfilling: Process, Technology, and Environmental Impact.* Christensen, T. H., Cossu, R. and Stegmann, R., Eds, Academic Press, London.

Tahir, M.; Hussain, T.; and Behaylu, A., 2015. “Scenario of Present and Future of Solid Waste Generation in India: A Case Study of Delhi Mega City.” Journal of

Environment and Earth Science, ISSN 2224-3216 (Paper), ISSN 2225-0948 (Online) Vol.5, No.8.

Taufiq, T., 2010. “Characteristics of Fresh Municipal Solid Waste.” M.S. Thesis, University of Texas at Arlington, Spring 2010.

TCEQ, Texas Commission for Environmental Quality.

Tchobanoglous, G.; Theisen, H.; Vigil, S. *Integrated Solid Waste Management, Engineering Principles and Management Issues*. McGraw-Hill, Inc.: New York, 1983.

Thompson, S.; Sawyer, J.; Bonam, R.K.; and Smith, S., 2005. “Review of Existing Landfill Methane Generation Model: Interim Report.” Natural Resources Institute, University of Manitoba, Winnipeg, Manitoba, Canada.

Tolaymat, T.M.; Green, R.B.; Hater, G.R.; Barlaz, M.A.; Black, P.; Bronson, D.; and Powell, J., 2010. “Evaluation of Landfill Gas Decay Constant for Municipal Solid Waste Landfills Operated as Bioreactors.” Journal of the Air & Waste Management Association, 2010, vol. 60, pp 91-97.

UNEP (United Nations Environment Program, 2001. “State of Waste Management in South East Asia.”

UNEP, 2005. “Closing of an Open Dumpsite and Shifting from Open Dumping to Controlled Dumping and to Sanitary Landfilling.” Training Modulus, United Nations Environment Programme.

US EPA, 2001. “RCRA Financial Assurance for Closure and Post-Closure.” Audit Report, 2001-P-007, US Environmental Protection Agency, Office of Inspector General, Washington, D. C., March 30, 2001.

US EPA, 2003. “Characterization of Municipal Solid Waste in United States.” *http://www.epa.gov/epaoswer/non-hw/muncpl/msw03.htm*

US EPA, 2005. "Landfill Gas Emissions Model (LandGEM)." Version 3.02 User's Guide, EPA-600/R-05/047, US EPA, Air Pollution Prevention and Control Division, Research Triangle Park, N.C., May 2005.

US EPA, 2007. "Bioreactor Performance." Office of Solid Waste, Municipal and industrial Solid Waste Management Division, EPA530-R-07-007.

US EPA, 2016. *http://www3.epa.gov/epawaste/nonhaz/municipal/index.htm.*

USDA, United States Department of Agriculture, Economic Research Service. *(http://www.ers.usda.gov/amber-waves/2013-june/ers-food-loss-data-help-inform-the-food-waste-discussion.aspx#.V3ClNvkrLIV)*

Vesilind, P. A.; Worrell, W. A.; and Reinhart, D. R., 2002. *Solid Waste Engineering*. Brooks/Cole Publishers, USA.

Viste, D. R, 1997. "Waste Processing and Biosolids Incorporation to Enhance Landfill Gas." Proceedings from Sardinia 97, 6th International Landfill Symposium, Vol. I, S. Margherita di Pula, Cagliari, Italy. I, pp 369-374.

Wang, Y.S.; Odle, W.S.; Eleazer, W.E.; and Barlaz, M.A., 1997. "Methane Potential of Food Waste and Anaerobic Toxicity of Leachate Produced during Food Waste Decomposition." Waste Management & Research, vol. 15, pp 149–167.

Warith, M. A. (2002). Bioreactor landfills: Experimental and field results. Waste Management 22 (1):7-17.

Warith, M. A.; Smolkin, P. A.; and Caldwell, J. G., 2001. "Effect of Leachate Recirculation on Enhancement of Biological Degradation of Solid Waste: Case study." Practice Periodical of Hazardous, Toxic, and Radioactive Waste Management, vol. 5 (1), pp 40-46.

Warith, M. A.; Zekry, W.; and Gawry, N., 1999. “Effect of Leachate Recirculation on Municipal Solid Waste Biodegradation.” Water Quality Research Journal of Canada, vol. 34 (2), pp 267-280.

Waste Disposal. World of Earth Science. 2003. Encyclopedia.com. 15 Feb. 2016.

Waste Management, Inc., 2002. “The Bioreactor Landfill: The next Generation of Landfill Management.”

World Bank, 2012. “WHAT A WASTE A Global Review of Solid Waste Management.” Urban Development Series Knowledge Papers. World Bank.

Wu, H.; Wang, H.; Zhao, Y.; Chen, T.; and Lu, W., 2012. “Evolution of Unsaturated Hydraulic Properties of Municipal Solid Waste with Landfill Depth and Age.” Waste Management, vol. 32 (3), pp. 463-470.

www.erosionpollution.com.

Yazdani, R., 2010. “Quantifying Factors Limiting Aerobic Degradation During Aerobic Bioreactor Landfilling and Performance Evaluation of a Landfill-Based Anaerobic Composting Digester for Energy Recovery and Compost Production.” PhD Dissertation, University Of California, Davis.

Yeny, D. and Yulinah, T., 2012. “Solid Waste Management in Asian Developing Countries: Challenges and Opportunities.” Journal of Applied Environmental and Biological Sciences., J. Appl. Environ. Biol. Sci., vol. 2(7) pp 329-335.

Yuen, S.T.S., 2001. “Bioreactor Landfills: Do They Work?” 2nd ANZ Conference on Environmental Geotechnics, Newcastle, Australia.

Zaman, M. N. B., 2016. “Effect of Manures on Food Waste Degradation in Biocell.” MS Thesis, University of Texas at Arlington, Fall 2016.

Zhang, R.; El-Mashad, H.M.; Hartman, K.; Wang, F.; Liu, G.; Choate, C.; and Gamble, P., 2006. “Characterization of Food Waste as Feedstock for Anaerobic Digestion.” Bioresource Technology, vol. 98 (4), pp 929–935.

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